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Effects of Fire Management of Southwestern Natural Resources

Proceedings of the Symposium

November 15-17, 1988
Tucson, AZ

Preface

Many changes have occurred in fire management in the southwest during the last decade. We have moved from a policy of "total" control to one where fire is used as a tool for resource management. A major impetus for this change in policy has been research findings that have provided us with a better understanding of the role fire plays in natural ecosystems, particularly southwestern plant communities. We know, for example, that fire is a useful tool for reducing fire hazards, improving wildlife habitat, and increasing water yield, to name just a few. In some cases prescribed fire is the only tool that exists since the use of forest chemicals has been questioned. But questions still keep surfacing regarding the specific effect of fire on the different resources. Likewise questions residing in the social-economic arena are less well known.

Fire research has been an important part of research in Arizona, New Mexico, and southwestern Colorado. Much valuable information on fire history has been gained, particularly in the ponderosa pine and mixed conifer types. The effect of fire on soils in chaparral, pinyon-juniper, ponderosa pine, and mixed conifer has been studies. We, the research community, have produced more good information on fire effects during the past 10 years than has resulted in all of past history. The need to make this information available to user groups led the Forest Service to propose and co-sponsor this symposium. The papers are designed to focus on the state-of-the-art effects of fire on several resources. The charge of the various authors was to synthesize what is known about the effects of fire on specific resources, and conclude with a section addressing management implications.

Although the papers describe many concerns surrounding the use of fire as a tool, many emerging issues still remain to be addressed. Many of these reside in the social, political, economic, and quality-of-life arena. Presentations at a recent fire meeting in California clearly indicated that the future fire problems in that state will be concerned about the effect of fire on urban-wildland interfaces, smoke management and air quality, and public health—along with their combined effect on the quality of life experienced by the state's residents. Although these problems are not yet as acute as in California, our rapidly growing population in the Southwest, particularly Arizona, will intensify these concerns within the next decade.

An extensive list of literature citations on fire effects relative to the southwest is included for your review. The citations provide references to literature on subjects not necessarily covered in this excellent symposium.

J. S. Krammes
Technical Coordinator

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The proceedings is a collection of papers and posters presented at the Symposium on Effects of Fire Management of Southwestern Natural Resources held in Tucson, Arizona, November 15-17, 1988. Included are papers, poster papers and a comprehensive list of references on the effects of fire on: plant succession, cultural resources, hydrology, range and wildlife resources, soils, recreation, smoke management, and monitoring techniques pertinent to prescribed fire management in the southwestern United States.

Keywords: fire management, prescribed fire, southwestern U.S., watershed resources, cultural resources, smoke management, monitoring.

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Effects of Fire Management of Southwestern Natural Resources

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Tucson, AZ

J. S. Krammes, Technical Coordinator

[1] Headquarters is in Fort Collins, in cooperation with Colorado State University. Krammes is at the Station's Research Work Unit in Tempe, in cooperation with Arizona State University.

Contents

(Continued)

Page

(Continued)

Role of Fire in the Management of Southwestern Ecosystems[1]

Henry A. Wright[2]

Abstract.—The role of fire varies greatly among plant communities depending on the kind, quantity, and continuity of fuel, fuel moisture, topography, and frequency of droughts. A wide variety of management problems plague us today because of a lack of fire in plant communities, but fire cannot be used to restore some of these communities without the use of hand, mechanical, or chemical treatment first. For many plant communities, fire can be used effectively today to achieve desired objectives if it is used by skilled technicians. Fire should play a greater role in the management of grass, shrub, and forest communities than it does today, although the use of fire is increasing.

The role of fire varies greatly among plant communities depending on kind, quantity, and continuity of fuel, fuel moisture, topography, and frequency of droughts. Because of lightning and a variety of human activities, fire has been a part of natural ecosystems since the origin of vegetation on earth. Historically, fire frequencies ranged from 2 to 400 years in most plant communities in North America, which created a wide range of fire intensities and varied vegetative mosaics (Wright 1987).

Without fire a number of undesirable things occur: excessive fuel buildups (i.e., Yellowstone Park), dense understories of shrub and trees in forests that could lead to catastrophic stand-replacing fires; decadent shrub and grassland communities; encroachment of shrubs and trees into grasslands; monocultures of trees that lead to increased disease and insect damage (i.e., lodgepole pine), as well as decreased wildlife diversity and, ultimately, uncontrollable fires.

Fire is not a cure-all for our management problems. However, it is effective and cheap for many land management problems if used by people skilled in the use of prescribed fire. In this paper I will highlight the historical role of fire and management implications for plant communities in the Southwest.

Semidesert Grass-Shrub

The historical role of fire in the semidesert grass-shrub community is somewhat perplexing. Photographs on the Santa Rita Experiment Station show grassland communities free of shrubs that today are dominated by velvet mesquite with grass understory. What caused the change? According to the early scientists at this experiment station, the change came about because of a lack of fire. My personal feeling is that occasional fires, in combination with drought, competition from herbs, rodents, and lagomorphs, played a significant role in controlling shrubs in the semidesert grass-shrub type. Also, the semidesert grass-shrub community is a delicate system (low rainfall and frequent droughts). Thus, I feel that grazing domestic livestock in combination with all the natural factors that might have kept much of the semidesert as a grassland, is too much for this system to tolerate and remain in its native state.

Management Implications

The southern desert grass-shrub type is a delicate ecosystem with wide swings in herbage yields because severe droughts are common. Moreover, droughts frequently last 2 or 3 years. When moderate to heavy grazing is imposed on black grama ranges, grass competition and vigor of grasses are drastically reduced (Canfield 1939). These factors favor high mortality of herbs during drought years and the eventual establishment of shrubs following wet years when other climatic factors, such as soil temperature, are favorable. It appears that use of fire would compound the existing problems on black grama ranges and may not have a place for shrub control on good ranges.

Our problem is to reclaim poor rangelands (predominately brush) and to properly manage our good rangelands. Poor rangelands cannot be managed with fire. These rangelands must first be restored using other reclamation techniques. However, once the rangelands are in good condition, fire can be used as an effective management tool in special situations during wet weather cycles to control burroweed, broom snakeweed, creosotebush, and young mesquite trees. Fires can also be used to suppress cactus species. Most burning should be done in June, but only following two previous years of better than average plant growth. This is especially important for grasses to recover quickly after burning.

Desirable shrubs that are either favored or not harmed by fire in-

[1] *Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 15-17, 1988).*

[2] *Horn Professor and Chairperson of Range and Wildlife Department, Texas Tech University, Lubbock, TX 79409.*

clude false mesquite, velvet-pod mimosa. Wright baccharis, and four-wing saltbush. Wheeler sotol and barrel cactus are easily harmed by fire and should be protected.

Today fire should be used only on a selective basis, or in combination with other methods, to achieve specific management objectives in the semidesert grass-shrub type. Fire probably has the greatest value to manage tobosagrass, sacaton, alkali sacaton, and mixed grama ranges. Good black grama grasslands appear to be too delicate to manage with fire. If fire is used, 3-4 years of rest might be required after a burn.

Arizona Chaparral

Throughout the world chaparral is thought of as a fire-induced vegetation type (Shantz 1947), although we know that much of the chaparral in Arizona is climax. Arizona chaparral burns periodically, but has a lower fire frequency than California chapparal. Burned chaparral areas that are left to recover naturally seldom support a reburn for at least 20 years, and many Arizona chaparral stands, particularly those that contain shrub live oak, are 80 to 100 years old (Cable 1975, Carmichael et al. 1978). In general, fire helps to keep chaparral communities diverse and productive if fires do not occur more frequently than 20 to 30 years. Many species (often palatable) are short-lived and must germinate from seed after a fire.

Management Implications

Fire will often improve the vigor of chaparral species as well as the diversity of species and nutrient values for wildlife. Shrub live oak, sugar sumac, skunkbush sumac, Wright silktassel, redberry, catclaw mimosa, Emory oak and yerbasanta resprout vigorously after fire. Other sprouters include true mountain mahogany, western mountain mahogany, and hair mountain mahogany. Pointleaf manzanita, Pringle manzanita, deerbrush, desert ceanothus, and Arizona cypress depend primarily upon seed for re-establishment. The two manzanita species, deerbrush, desert ceanothus, yerbasanta, and yellowleaf silktassel are considered fire-induced species (Carmichael et al. 1978). Grasses and forbs appear prevalent through the fourth year and disappear by the fourth year. Fire may not be necessary in some chaparral communities, but may be needed every 30 to 50 years in dense stands of chaparral. Removal of dense stands of chaparral followed by the seeding of grasses on slopes less than 30% will enhance overland flow to reservoirs and municipalities.

Oak Brush

The oak-brush zone is just above the pinyon-juniper one of the central and southern Rocky Mountains. Often it is an understory species in ponderosa pine, particularly in northern Arizona and New Mexico. Fire frequency in the oak-brush zone is 50 to 100 years, although this is speculative. Most fires in Gambel oak occur after a buildup of litter and mulch under the shrub mottes and generally occur during dry periods. Most fires are spotty and irregular.

Management Implications

Gambel oak is very tolerant to fire, and fire does not appear to improve herbage yield or species composition, unless the areas are reseeded to introduced grasses. For the most part, these areas should be left untreated to stabilize soil, retard snowmelt, and provide browse for deer. Accidently burned areas have been seeded with intermediate wheatgrass, smooth brome, and fairway wheatgrass, which increased grass yields and reduced oak growth.

Pinyon-Juniper

Historically fire has been the dominant force controlling the distribution of pinyon-juniper, particularly juniper, but fire cannot be separated from the effects of drought and competition. All three forces seem to have played a complementary role in limiting the distribution of juniper before grazing by domestic livestock was a factor. However, droughts and competition from grass probably only served to slow the invasion and growth of junipers in adjacent grasslands, since the trees are easily established during wet years (Johnsen 1962), especially where shade is present (Meagher 1943). Fire, occurring about every 10 to 30 years (Leopold 1924), kept the junipers restricted to shallow, rocky soils and rough topography (Arend 1950, Burkhardt and Tisdale 1969, O'Rourke and Ogden 1969). For the last 70 years, however, heavy livestock grazing has reduced grass competition as well as fuel for fires. Reduced competition from grasses has permitted pinyon and juniper to invade adjacent communities rapidly (Nabi 1978) and the reduced number of fires, each of a lower intensity than the fires before heavy grazing, has left the juniper invasion unchecked.

Management Implications

In open stands of pinyon-juniper in the Southwest, fire can be used effectively to kill pinyon and juniper trees less than 1.2 m (4 ft) tall. Taller trees are very difficult to kill, even with hot fires, unless tumbleweeds have accumulated at the tree bases. Open stands of tall pinyon and juniper trees are not considered undesirable, but can be eliminated by chaining or dozing followed by burning, to render the microclimate unfavorable for tree seedlings.

Several management agencies have tried various techniques (Arnold et al. 1964, Aro 1971) to reclaim

closed pinyon-juniper stands (with no understory of grasses or shrubs). Prescribed burning, or some combination of burning with other treatments (followed by artificial seeding when necessary) is the most effective procedure to reclaim closed stands of pinyon-juniper (Aro 1971, Springfield 1976). Without any prior treatment, burning must be done on hot days [35° to 38°C (95° to 100°F)] with low relative humidity and 13- to 32-km/hr (8 to 20-mi/hr) winds, conditions considered too hazardous by most land managers (Arnold et al. 1964). Thus mechanical treatment followed by burning is probably the most acceptable technique to reclaim dense stands of pinyon-juniper, even though it is expensive. Burning should be delayed 2 to 3 years after chaining to assure that most of the pinyon and juniper seeds have germinated.

Grasses will increase dramatically following burning and seeding treatments in closed stands of pinyon-juniper. Herbage yields on the Hualapi Indian Reservation in northern Arizona, seeded with crested wheatgrass, western wheatgrass, weeping lovegrass, and yellow sweetclover, produced 1865 kg/ha (1650 lb/acre) after treatment compared with 65 kg/ha (60 lb/acre) for the unburned control (Aro 1971). On another large-scale burning and seeding program in pinyon-juniper woodland, Aro (1971) reported that forage production increased 560 kg/ha (500 lb/acre). Pinyon-juniper communities in northern Arizona that were chained and seeded but not burned produced 1100 kg/ha (980 lb/acre) of grasses, forbs, and shrubs 5 to 11 years after treatment, compared with 250 kg/ha (220 lb/acre) on control plots (Clary 1971). Where native grasses were present in the understory, reseeding was not necessary (Aro 1971).

Mixtures of sagebrush and pinyon-juniper can be burned without prior treatment. Generally thick stands with 45% to 60% cover are selected for burning and burned into areas with less shrub cover. Some areas are left to reseed naturally, but aerial seeding is usually considered desirable.

Pinyon-juniper stands converted to grassland should be reburned about every 20 to 40 years. A definite time is difficult to set because reinvasion of pinyon and juniper is dependent on the kind of initial treatment, time span between treatments, intensity of the burn, and grazing intensity after the burn. A better guide would be to reburn when the average height of pinyon or juniper reaches 1.2 m (4 ft).

Ponderosa Pine

Historical evidence indicates that fires have always been an ecological force in ponderosa pine communities, regardless of whether they were seral or climax. In Arizona and New Mexico, the frequency of natural fires in climax and seral ponderosa pine communities varied between 4.8 and 11.9 years (Weaver 1951). Fires thinned the stands, eliminated young pines and/or climax mixed-conifer species including thickets, and kept the ponderosa pine forests open and parklike with an understory of herbs and shrubs (Biswell 1972, Cooper 1960, Hall 1976, Weaver 1947).

Management Implications

Many of our ponderosa pine forests are overstocked, stagnate, or have dense understories of young trees under mature trees. These stands pose serious management problems. They were created because of lack of fire, but can rarely be restored with fire as the initial treatment. Expensive handwork along with special equipment and pile burning may be necessary to restore many stands of ponderosa pine to a productive state that can be maintained with fire. Management of ponderosa pine on the Fort Apache Reservation in Arizona gives us a good example of how we can use fire in our national forests and keep them productive.

Fire should play a much greater role in the management of our ponderosa pine forests than it does today, although it is not equally necessary in all associations that support ponderosa pine. It will take much time to get ponderosa pine forests to the stage where they can be easily managed with fire. But once that stage is achieved, fire hazards and suppression costs will be reduced and investments for cultural treatments will decline yet forests will be both more productive and attractive.

Mixed Conifer

Most mixed conifer stands in the southern Rocky Mountains were established after fire (Moir and Ludwig 1979). Arizona and New Mexico have the highest frequency of lightning fires in the United States and Canada (Schroeder and Buck 1970). The drier, lower elevation habitat types dominated in climax by white fir, Douglas-fir, and ponderosa pine and in seral stages by ponderosa pine burned about every 5 to 12 years (Weaver 1951). On cooler, moist sites such as the white fir-Douglas-fir/boxelder habitat type, the mean fire-return interval would be longer. In the White Mountains of Arizona, Dieterich (pers. comm.) estimated the fire-return interval to be 22 years. Fires would be either light and erratic due to wet fuels or intense stand-replacing fires during dry years because of the heavier fuel loadings.

Management Implications

Before Europeans arrived in America, ground fire was most common in the southern Rocky Mountains with the occasional crown fire where

there was a combination of heavy fuels and extreme fire weather conditions. Today many stands have not been burned for over 100 years and are vulnerable to major fires such as those that occurred in Yellowstone Park this year. In dry, windy weather, crown fires will kill most trees because of heavy fuel accumulations and the continuous ladder of fuel from ground to treetop. Rather than to let nature take its course during the dry years, we should be doing prescribed burning where we have our heaviest downed fuel loads and burn under conditions to achieve desired objectives. Otherwise, major stand-replacing fires will be more common than they are now, and reestablishment through natural seeding will be a problem in many areas.

Shrub-Grasslands in Texas

There are not reliable historical records of fire frequencies in the southern mixed prairie of the Great Plains grassland because there are no trees to carry fire scars from which to estimate fire frequency. However, we estimate that fires occurred every 5 to 10 years because topography is level to rolling in the Great Plains and explorers and settlers were always concerned about the danger of prairie fires (Wright and Bailey 1982). Based on my experience in shrub-grasslands, fires occur most often during droughts that follow 1 to 3 years of above average rainfall when fuel is abundant. These fires suppress shrub growth and provide herbaceous competition for new shrub seedlings. Moreover, rodents and lagomorphs feed on the young seedlings and resprouts for metabolic moisture during all seasons of the year, which will keep the shrubs weakened as long as their abundance is below the threshold level that can be used by rodents and lagomorphs. Intermittent droughts further complicate the ability of shrub seedlings to establish and for shrubs to resprout.

Management Implications

Fires have played a very important role in the Southern Great Plains to control the cover of shrubs. With prescribed burning we can use fire to effectively suppress honey mesquite, redberry juniper, Ashe juniper, and cactus species. Other benefits include increased utilization of coarse grass species, a shift in composition of forbs and shrubs that enhance wildlife habitat, removal of dead wood on the ground, and short-term control of undesirable forbs and annual grasses. Increased visibility for gathering livestock is another major benefit to ranchers.

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Fire History and Climate in the Southwestern United States[1]

Thomas W. Swetnam[2]

Abstract—Forest fire occurrence during the past three centuries was examined using historic records from documents and fire-scarred trees. The influence of climate on fire regimes was apparent in regional synchroneity of large fires and association of reduced fire activity and El Niño-Southern Oscillation (ENSO) events. The latter association may have forecasting value.

In the study of centuries-long patterns of change in forest ecosystems one can observe recurrent themes, even "cycles," if we define this term in a nonclassical sense. For example, consider the well known successional pathway of aspen stands establishing after stand-replacing fires in mixed-conifer or spruce-fir forests. The aspen are, in turn, replaced by regenerated shade-tolerant conifers and, finally, recurrence of another fire storm may repeat the process. Management practices on public lands also seem to run in cycles. Consider the historical fact that fire suppression was essentially nonexistent in western forests before the first decade of this century. "Let burn" practices held sway in a number of areas in the 1910s and 20s, particularly California. The "hit-em hard and hit-em fast" strategy of the 10 AM policy finally won out completely in a heated controversy in the early 1930s, and the pendulum was then solidly on the total suppression side. In the 1970s and 80s land management agencies in the western U.S. shifted back toward the approach of using fire for management purposes. Although the term "let-burn" was still banned from the fire management lexicon, it was clear that the pendulum had moved back toward center.

Part of the cycling of fire policy may be linked to the cycling of fire regimes. Public (i.e., political) support of the early total suppression policy grew out of the shock of devastating fires in the late 1800s, such as the Hinckley and Peshtigo fires in the Lake States, and the 1910 fires of the Northern Rockies. Now, following the catastrophic fires in California and the Pacific Northwest last year, and the Yellowstone and other western fires this year, we are seeing a similar "fire shock" both in the public and land management agencies. Are we now poised for another shift in fire policy, presumably back toward the total suppression side? I think not, but any reassessment of policy has much to gain from a historical perspective. If we are to avoid further "fire shocks" we must recognize that regional-scale fire years have occurred in the past, sometimes despite aggressive total suppression policies, and these events will very likely recur in the future. Management strategies must be geared to minimizing these resource impacts through forecasting of hazardous conditions, fire–fighting readiness, and prudence in the use of prescribed fire.

The main theme of this paper is that historical and ecological knowledge can provide managers with perspectives necessary to meet these challenges. The focus is on historical records of fire and climate in Arizona and New Mexico during the last 300 years. Analyses of weather records also reveal that one particular climatic phenomena, the El Niño-Southern Oscillation (ENSO), seems to explain a substantial amount of the year-to-year fire load in the southwest. The ENSO-Southwestern fire teleconnection (Swetnam and Betancourt, in preparation) suggests that during some years it may be possible to forecast fire season severity several months in advance.

Methods

Historical fire records were compiled from two general sources: (1) Forest Service documents covering the post-1900 period, and (2) fire-scar records on trees covering the pre-1900 period. The Forest Service documents included data on number of fires and acres burned per year in the Southwestern Region (Arizona and New Mexico), USDA Forest Service. These data were analyzed by size class and totals. Sources were annual National Forest Fire Reports (USDA, Forest Service) and a detailed study of southwestern lightning fire data conducted by Barrows (1978). A longer record was compiled for the Gila National Forest in southwestern New Mexico from fire atlases and individual fire reports (Swetnam 1983a), and from an early fire statis-

[1]*Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Assistant Professor of Dendrochronology, Laboratory of Tree-Ring Research, University of Arizona, Tucson, AZ 85721.*

tics study by E. W. Loveridge in 1926 (files at Gila National Forest Supervisor's Office, Silver City, NM).

Pre-1900 fire occurrence data were compiled from fifteen fire-scar chronologies. These include 13 chronologies from Arizona and New Mexico, one from Mexico (Sierra Los Ajos, Sonora) and another from Texas (Guadalupe Mountains National Park) (fig. 1.). The fire-scar chronologies were developed by collecting full or partial cross section samples from living and dead fire-scarred trees, crossdating the annual rings (Stokes and Smiley 1968) and observing the position of fire scars within the annual rings (Dieterich and Swetnam 1984). Table 1 lists published and unpublished sources of the data sets and various other characteristics of the fire regimes.

Results

Post-1900 Fire History

A positive trend in number of fires reported each year was evident in the records of fire occurrence. Most of this effect was due to improvements in detection and fire-fighting resources available to send to fires. For example, in the Gila National Forest there were only 4 to 6 fire guards on horseback available for fire fighting from about 1909 to 1933, and the number of fires reported was relatively low during this period with no noticeable upward trend (fig. 2). Then in 1934 over 100 men of the Civilian Conservation Corps (CCC) were stationed within the National Forest and the fire record shows a obvious upward trend. Other detection and fire suppression improvements also correspond to increases in total numbers of reported fires (fig. 2). Most of this increase, however, was in the number of small fires (class A, 0.25 acres or less). It seems likely that in the first 20-year period the Gila fire guards

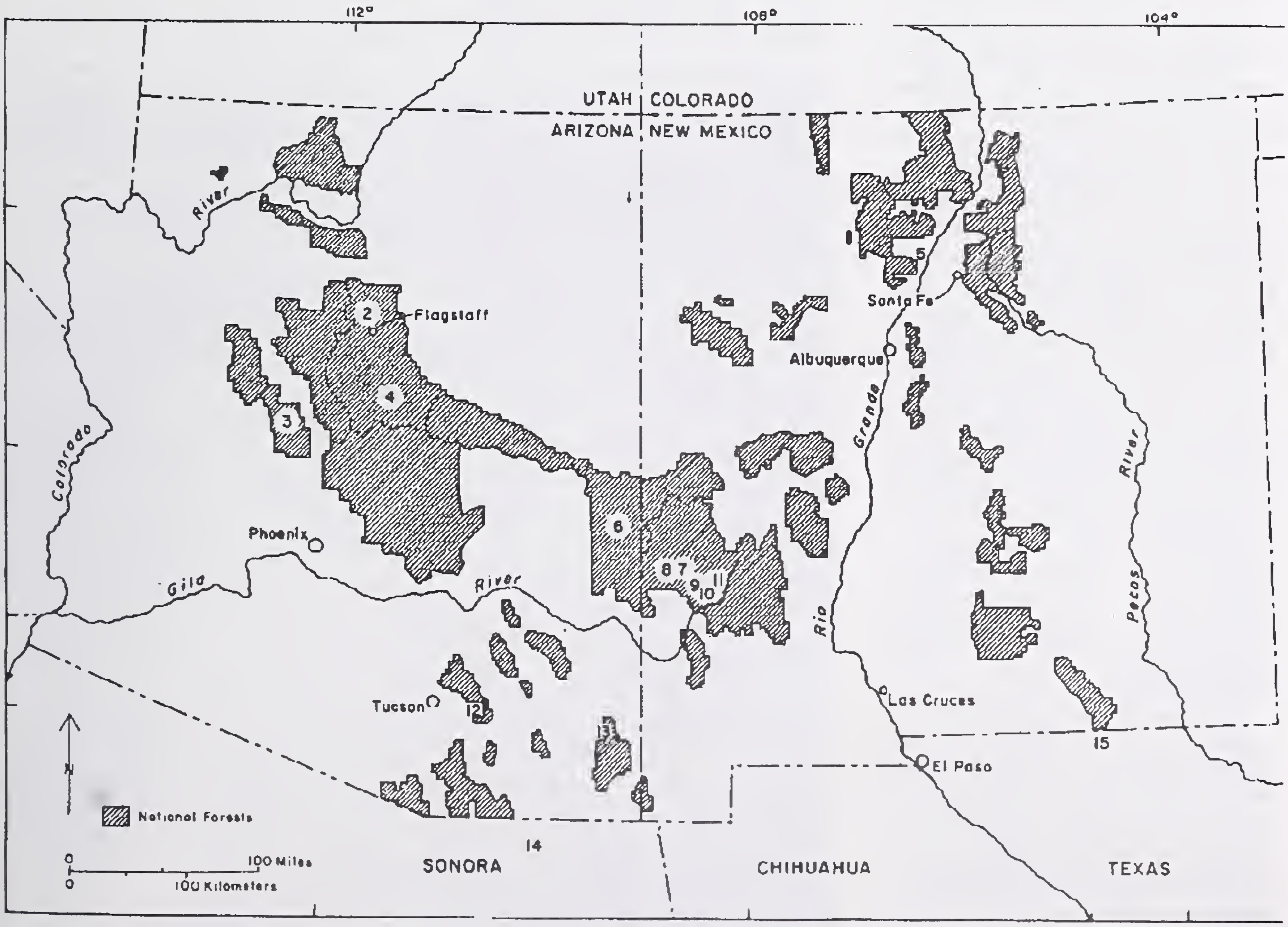

Figure 1.—National Forests in the Southwestern Region, U.S. Forest Service, Arizona and New Mexico. Fire scar collections are numbered and are described in table 1.

were probably unable to detect all fires that started, and even if they had they would not have been able to go to all of them.

Although improvements in detection and increase in manpower explain most of the increase in fire numbers in the Gila, there also appears to be more larger fires. The number of large fires reported in the Southwestern Region may also have a slightly upward trend (fig. 3). Table 2 compares the number of fires by size class for selected early and late periods in the Southwestern Region and the Gila. For the Southwestern Region, only the number of class A fires has significantly increased in the modern period; however, the number of fires in all size classes has increased in the Gila.

While improved detection and fire-fighting resources may account for the increase in number of small fires the increase in number of larger fires, may have a different explanation. Possibilities include change in climate and/or change in fuel loadings toward conditions favoring larger fires. Another possible contributing cause may be increased commercial and public use and access to National Forest lands, because there was also a statistically significant increase in average annual number of person-caused fires in the modern period (Gila record: 1909-1939, $X = 12$; 1960-1986, $X = 18$, $t = 2.87$, $p = 0.006$) (also see fig. 2). It is not known how much of the increase in large fires in the Gila was due to human ignitions because data on person-caused fires classified by size were not obtained.

Other evidence suggests that an increasing trend in fire load was not related to person–caused fires. Consider the lightning fire data for the Southwestern Region (table 3).

Barrows (1978) compiled the first 36 years of this record, and he also noted the steady rise in area burned per fire. He pointed out that the high area burned figure for the 1970s was strongly influenced by large fires in 1971 and 1974; 1979 was also an unusually heavy fire-load year.

One possible bias in the record after about 1974 was the increased area burned under prescribed natural fire programs, where lightning ignited fires were allowed to burn during special conditions. However, when the area burned under this program in the Gila Wilderness (which had the most active program well into the 1980s) was discounted from the 1970-1979 period, the average area burned per lightning-caused fire was still 6.66 ha, which was more than 40% greater than the previous 10-year period. The magnitude of the prescribed natural fire effect on area burned per fire for the 1980-1987 period has not yet been assessed because of lack of data. In any case, the rise in area burned per lightning-caused fire from 1940 to at least 1979 seems to be a genuine phenomenon that suggests a worsening fire situation in the Southwestern Region.

Obvious trends in total area burned per year were generally not observed (fig. 4). However, large peaks during certain years of this record are notable, reflecting especially severe fire years (e.g., 1925, 1946, 1951, 1956, 1959, 1960, 1971, 1974, 1979, and 1985). In some years, large burned area values can be traced to one or a few very large fires on a single National Forest, such as 1951 when two fires burned over 20,000 ha in the Gila. However, the majority of the peaks in fire activity in the Southwestern Region record represent regional-scale fire years

Table 1.—Fire-scar chronologies from southwestern United States. Periods of sample coverage between 1700 and 1900 were selected for comparsion of fire interval ststistics among the sites. Fire intervals were periods in years between fire events that were recorded by more than 10% of sampled trees that were fire-scar susceptible at the time of each fire event. Numbers beside site names refer to figure 1.

Site	Name	Forest[1] type	Period	Fire intervals (years) Mean	S. Dev	Max.	Source[2]
1.	Chuska Mountains	P	1700-1895	5.0	5.2	24	1
2.	Chimney Spring	P	1754-1876	2.5	1.5	8	2
3.	Battle Flat	P	1750-1861	1.9	1.5	11	3
4.	Limestone Flats	P	1750-1998	2.9	2.0	10	4
5.	Frijoles Canyon	P-MC	1709-1899	7.3	5.1	23	5
6.	Thomas Creek	P-MC	1702-1893	9.8	6.9	26	6
7.	Gilita Ridge	P	1705-1899	4.9	3.1	18	7
8.	Bearwallow	MC	1705-1879	7.3	5.5	21	8
9.	Langstroth Mesa	P	1705-1892	5.1	3.8	22	7
10.	McKenna Park	P	1705-1890	4.5	2.8	12	7
11.	Black Mountain	P-MC	1702-1899	6.5	4.1	20	9
12.	Mica Mountain	P-MC	1703-1893	6.1	2.6	13	10
13.	Rhyolite Canyon	P-MC	1707-1886	8.5	5.5	21	11
14.	Sierra Ajos	P	1800-1899	4.5	2.1	11	12
15.	Guadalupe Mtns.	MC	1704-1879	8.8	6.9	26	13

[1]*Forest Type: P = Pure or nearly pure ponderosa pine forest; MC = mixed conifer forest; P-MC = Mixed conifer forest but with dominant component of ponderosa pine in some areas and other species such as Douglas-fir and white fir also present.*

[2]*Sources (unpublished reports and data sets on file at Laboratory of Tree-Ring Research, University of Arizona, Tucson, are indicated with *): 1. *Savage (1988); 2. Dieterich (1980a); 3. Dieterich and Hibbert (this volume); 4. Dieterich (1980b); 5. *Caprio et al. (1988) and Allen (1988); 6. Dieterich (1983); 7. Swetnam and Dieterich (1985); 8. *Baisan (1988b); 9. *Baisan (1988a); 10. *Baisan (1988b); 11. *Swetnam et al. (1988); 12. *Swetnam (1983); 13. Ahlstrand (1980).*

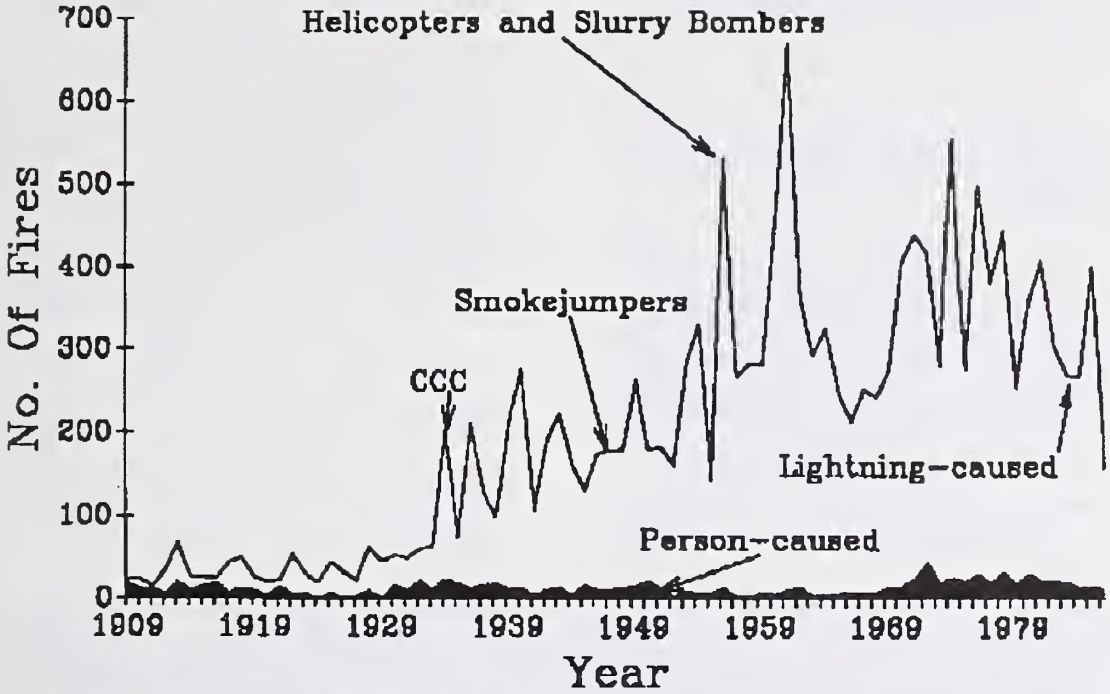

Figure 2.—Number of fires reported per year, 1909-1986, in the Gila National Forest, New Mexico. Approximate timing of improvements in fire-fighting technology correspond with increases in numbers of reported fires.

when large fires were reported in many of the National Forests.

Pre-1900 Fire History

Time series of the fifteen fire-scar chronologies listed in table 1 are illustrated in figure 5. This chart demonstrates the frequent and aperiodic nature of forest fires in ponderosa pine and mixed-conifer forests of the Southwestern Region during the two centuries before 1900. One of the most striking and consistent observations derived from nearly all of these histories was the sudden end of a surface fire regime in the late 1800s, or very soon after the turn of the century. Very few or no fire scars were recorded on any of the trees represented in figure 5 after 1900, with the exception of the chronology from Sierra Los Ajos. This chronology recorded a relatively uninterrupted regime of frequent surface fires extending into the 1980s when the trees were sampled. Recurring fires in the twentieth century at this site may reflect few or no attempts to suppress fires on this relatively remote mountaintop of northern Mexico.

Many fires during the presuppression era burned for months at a time, and they covered thousands of hectares. We know this must be true because lightning ignites fires in some ponderosa pine forests as early as April, with at least two months of dry conditions remaining. Some fires might have persisted through the rainy season of July and August and flared up again during drier periods of late summer and early fall. Also, fire scar records collected from numerous trees scattered over ponderosa pine forests as large as 2,000 ha show matching fire dates among most trees for most fires, indicating very large burns were typical in this type.

The mean fire intervals for all chronologies range between approximately 2 and 10 years (table 1). Mean fire intervals in ponderosa pine sites on the Mogollon Rim (central Arizona and west-central New Mexico) and southward range from 4 to 5 years, while mean fire intervals at higher elevations in mixed-conifer types range from 6 to 10 years. The mixed-conifer chronologies often recorded smaller fires at short intervals. When only larger fires in this type were considered, the mean fire intervals were on the order of 15 to 25 years (e.g., Dieterich 1983). Three of the northern Arizona chronologies (Limestone Flats, Chimney Spring and Battle Flat) recorded the highest fire frequencies of any forest type, with mean fire intervals ranging between 1.9 and 2.9 years. Consecutive-

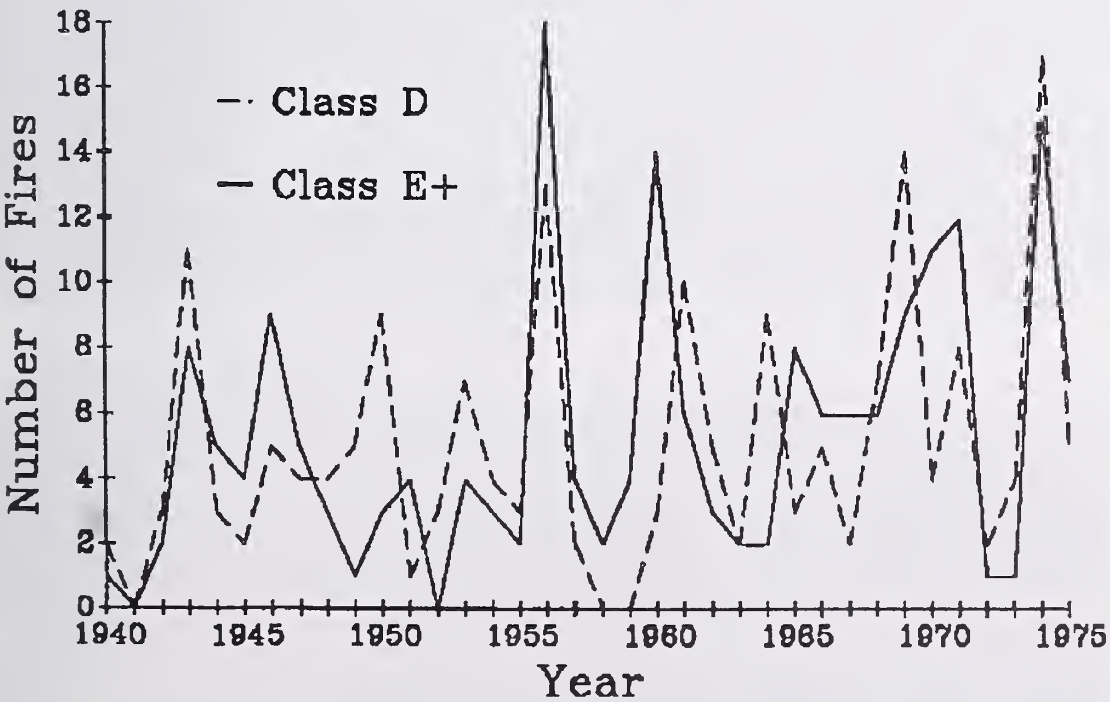

Figure 3.—Number of class D (100 to 299 acres) and class E and larger (> 300 acres) lightning fires in the Southwestern Region (Barrows 1978).

year fires (1-year intervals) have also been noted on rare occasions in some of these sites. The higher fire frequencies in northern Arizona chronologies, with exception of the Chuska Mountains, is clearly visible in figure 5.

The mean recurrence intervals shown in table 1 are primarily useful for comparative purposes. It is most important to consider the historical range and variability of fire occurrence. One example of temporal variability can be seen as an apparent shift in fire regimes to longer fire-free periods beginning in the early and middle 1800s in many of the chronologies (fig. 5). A puzzling gap with few or no fires during a period from about 1820 to 1840 is also noticeable in some of the chronologies. This hiatus may be due to wetter conditions as evidenced by climate reconstructions from tree rings; the 1830s-40s period was apparently one of the wettest in the last two hundred years (Schulman 1956, Fritts 1986). Also, reduction in fire frequency during this period may have been associated with intensified sheep grazing in New Mexico during the 1820s (Denevan 1967). Intensive grazing removes fine grassy fuels important in spreading fires. However, information about mid-19th century stock numbers in southern New Mexico and Arizona is sketchy. The grazing hypothesis may be supported by the fact that many of the fire chronologies recorded a final fire in the late 1800s around the beginning of an intensive grazing era, and 20 years or more before the advent of organized fire suppression efforts by the USDA Forest Service.

The fire scar chronologies also show a remarkable correspondence of fire years across the Southwest (fig. 5). These years are also evident in a fire area index (fig. 6). The fire area index reveals that regional-fire years (labeled years) were often large fires within the sites where they were recorded. The individual chronologies (fig. 5) show that in some cases more than 80% of sampled trees recorded fires during the regional-fire years.

Southwestern Fire and ENSO Events

El Niño-Southern Oscillation (ENSO) events are global-scale climatic anomalies that recur at intervals of 2 to 10 years and at varying intensities (Philander 1983). The El Niño pattern is characterized by weak tradewinds and appearance of high sea-surface temperatures off the western coast of the Americas. The Southern Oscillation is measured as the normalized differences in monthly mean pressure anomalies between Tahiti, French Polynesia and Darwin, Australia. El Niño and the Southern Oscillation are linked in a global climate complex of changing ocean currents, ocean temperatures, atmospheric pressure and temperature gradients. Subcontinetal or regional-scale climatic effects of ENSO events are highly variable. In some cases the effects are consistent and extreme, leading to droughts in some regions and flooding in others.

Table 2.—Changes in annual number of fires by size class (acres). Southwestern Region data include only lightning-caused fires, while Gila National Forest data include lightning and person-caused fires.

	A (0.25 or less)		B(0.26-9)		C+ (10+)	
	Mean	SD	Mean	SD	Mean	SD
Gila National Forest: 1.4 million hectares						
Period						
1909-1939 (31 yrs.)	40	43	16	12	5	5
1960-1980 (21 yrs.)	276	103	79	35	10	10
t-statistic =	11.370		9.190		2.790	
probability =	< 0.001		< 0.001		0.008	
Southwestern Region: 8.4 million hectares						
Period						
1940-1957 (18 yrs.)	1,063	329	324	100	47	22
1958-1975 (18 yrs.)	1,515	437	307	110	51	31
t-statistic =	3.505		0.460		0.459	
probability =	0.001		0.646		0.649	

Table 3.—Lightning fire data for the Southwestern Region.

Period	No. of fires	Hectares (acres) burned	Hectares (acres) burned/fire
1940-1949	13,858	49,982 (101,266)	2.96 (7.31)
1950-1959	15,709	53,358 (131,846)	3.40 (8.39)
1960-1969	17,363	67,882 (167,733)	3.91 (9.66)
1970-1979	19,925	136,791 (338,006)	6.86 (16.96)
1980-1987	10,716	61,179 (151,171)	5.71 (14.11)

A possible link between ENSO related droughts in the western Pacific and fire is evidenced by massive bush fires in Australia during the 1939 and 1982-83 El Niño events. Also, one of the largest forest fires in history occurred in the tropical forests of the Kalimantan province of Indonesia in 1982. This gigantic fire is estimated to have burned over 3.1 million hectares!

Simard et al. (1985) compared ENSO events to 53-year fire records from large regional areas of the United States and found that only the southeastern region had a statistically significant relationship (inverse) with this climate phenomena. In their study, Arizona and New Mexico were grouped with all the Rocky Mountain and Great Basin states northward to the Canadian border. They noted, however, that this comparison was probably at too coarse a scale, and that smaller and more detailed regional studies may identify stronger relationships.

In the southwestern United States, ENSO events are most consistently related to wetter than average spring and fall seasons (Andrade and Sellers 1988, Betancourt 1988). This condition is due to southward displacement of the jet stream and westerlies in the eastern Pacific and development of tropical storms over the warming waters off the coast of Baja, California. Subsequent anti-cyclonic movement of these storms into northern Mexico, southern Arizona and New Mexico brings an unseasonable influx of moisture to the region during the normally arid spring and fore-summer. This is also the critical burning period in the Southwestern Region (fig. 7).

Figure 8 shows the timing of ENSO events as classified by Quinn et al. (1987) and area burned per year in the Gila National Forest and the Southwestern Region. Differences in mean hectares burned per year between ENSO and non-ENSO years were evaluated using the Mann-Whitney test (table 4). The ENSO event in 1925 appears to be the only outstanding exception to a consistent correspondence of ENSO events and the lowest area burned years (fig. 8). Unlike most ENSO events, the spring of 1925 was actually quite dry, and wetter conditions did not develop in the Southwest until the fall and winter of 1925-26. Note that one of the most severe ENSO events of the past two centuries occurred in 1982 and 1983, when the area burned in the Southwest was the second and third lowest over the past 50 years. The lowest area burned was in the year 1941, another very severe ENSO event.

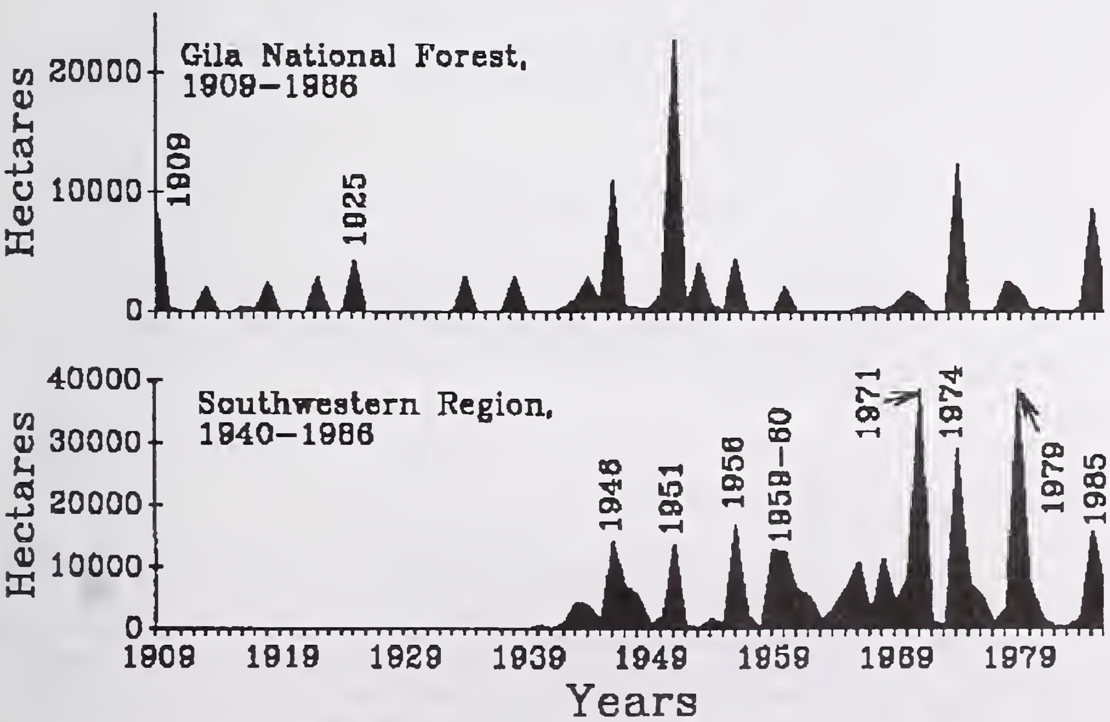

Figure 4.—Area burned per year in Gila National Forest and Southwestern Region. The Gila data include person and lightning-caused fires, while Southwestern Region data include only lightning fires. Major fire years are labeled.

Initial observations of number of fires recorded per year in relation to ENSO events has yielded inconsistent results. Simard et al. (1985) also noted that these types of data were less clearly related to ENSO than area burned. This may be due to the confounding effect of changing fire-detection and fire-fighting resources. A more complex relationship may also exist with thunderstorm activity and ENSO. For example, increased moisture during some ENSO years may actually result in increased thunderstorm activity which triggers more lightning caused fires, but lower overall area is burned because of higher fuel moisture. Weak or moderate El Niño events also seem to be less clearly associated with reduced southwestern fire activity than severe or very severe events. Again, it seems possible that ignition rates and fuel moisture may be involved. Weak or moderate El Niño events may deliver enough moisture to the Southwest to increase the level of thunderstorm activity but not sufficient moisture to reduce fire spread.

The next ENSO fire comparison utilized the much longer-term perspective of fire-scar chronologies. The two–century record of fire occurrence derived from the five Gila fire-scar chronologies was compared to severe and very severe ENSO events in figure 9. El Niño events for the period prior to the mid-nineteenth century were reconstructed from historic records of droughts, floods, fluctuations in fisheries off the South American coast, and other variables that are known to be closely linked to the El Niño pattern (Quinn et al. 1987).

With only a few exceptions the severe ENSO events corresponded to years of reduced fire activity as measured by fire-scarred trees (fig. 9). Another fairly consistent pattern was occurrence of large fire years one or two years following severe ENSO events (fig. 9).

Discussion: Implications for Fire Management

Trends and Extreme Events

Two particular warnings to managers emerge from the study of fire records. The first is a trend of increasing fire load. Evidence of this trend includes higher numbers of large fires in the Gila National Forest in the modern period and larger area burned per lightning-caused fire in the shorter record from the entire Southwestern Region. This trend may indicate that fire control will be an increasingly difficult task in coming years. Possible causes of this trend may be climate change, management practices, or both.

Management practices might promote large fires in several ways. The increased number and extent of roads in southwestern forests provides more access to forest lands. Improved access coupled with increased recreation use beginning in the 1970s may have led to more person-caused fires. Increased fuels and changes in forest structure due to 80 years of relatively effective fire control, especially in pine forests which formerly burned 2 or 3 times per decade, may also have played a role in the apparent increasing trend in lightning fire data. Logging also increases fuel loadings on the forest floor in some types, and opening of canopies in harvested stands might lead to more rapid drying of fuels because of greater insolation.

The second major warning to managers from the fire data analysis is that extreme fire events, or regional-fire years, are part of the fire

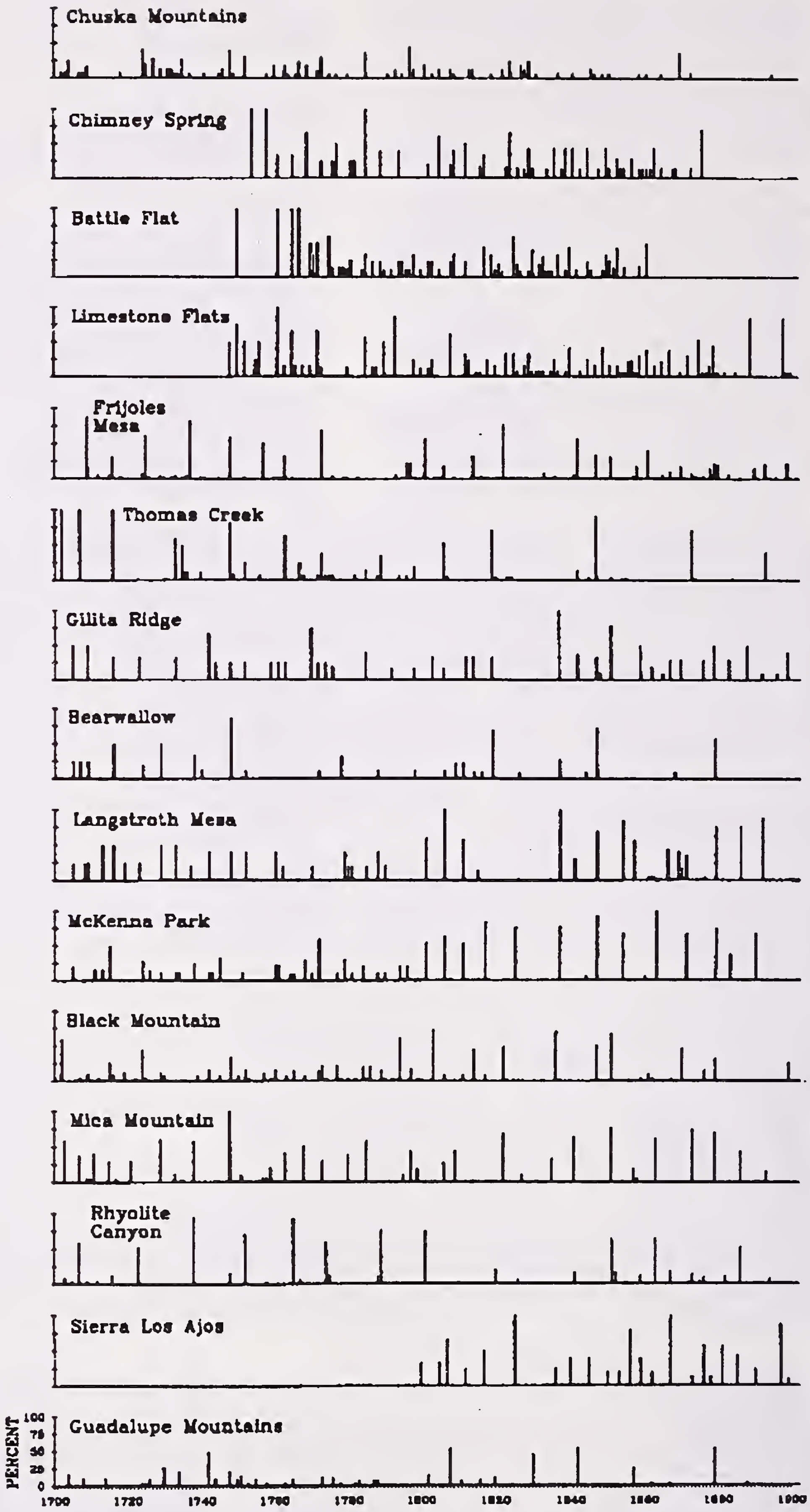

Figure 5.—Fire-scar chronologies from the Southwest. Percent trees scarred is the proportion of sampled trees recording a fire relative to the number of fire-scar susceptible trees at that date. Fire-scar susceptible trees are defined as trees that have been previously scarred by at least one fire (Romme 1980). The wide range of variability in fire chronologies is visible. Notice the higher fire frequency in chronologies from northern Arizona (four uppermost plots), and the longer intervals between fires that appear in 1800s in the chronologies further to the south.

regime of southwestern forests. This is not particularly new information to southwestern fire–fighting veterans who remember, for example, the 1956, 1971 and 1974 fire seasons. Nevertheless, consideration of history extending beyond the reach of human memories can serve to verify the existence of this phenomenon, and possibly help identify the probability of its recurrence.

One of the remarkable features of the two-century southwestern fire history represented in the fire-scar chronologies was the synchrony of large fire years among the widely scattered sites. Based on both the Forest Service data base for this century and the fire-scar record extending back to 1700, it appears that at least 3 or 4 regional-fire years occurred per century. The occurrence of extreme regional-fire years suggests that atmospheric conditions were responsible, because these are the only environmental variables that can account for such spatially large-scale phenomena. It is also likely that these conditions were at least partly cumulative through some period of time preceding extreme fire events, and therefore some advance warning may be present. There is clearly a need for further research of historical climate and fire records to pinpoint these conditions.

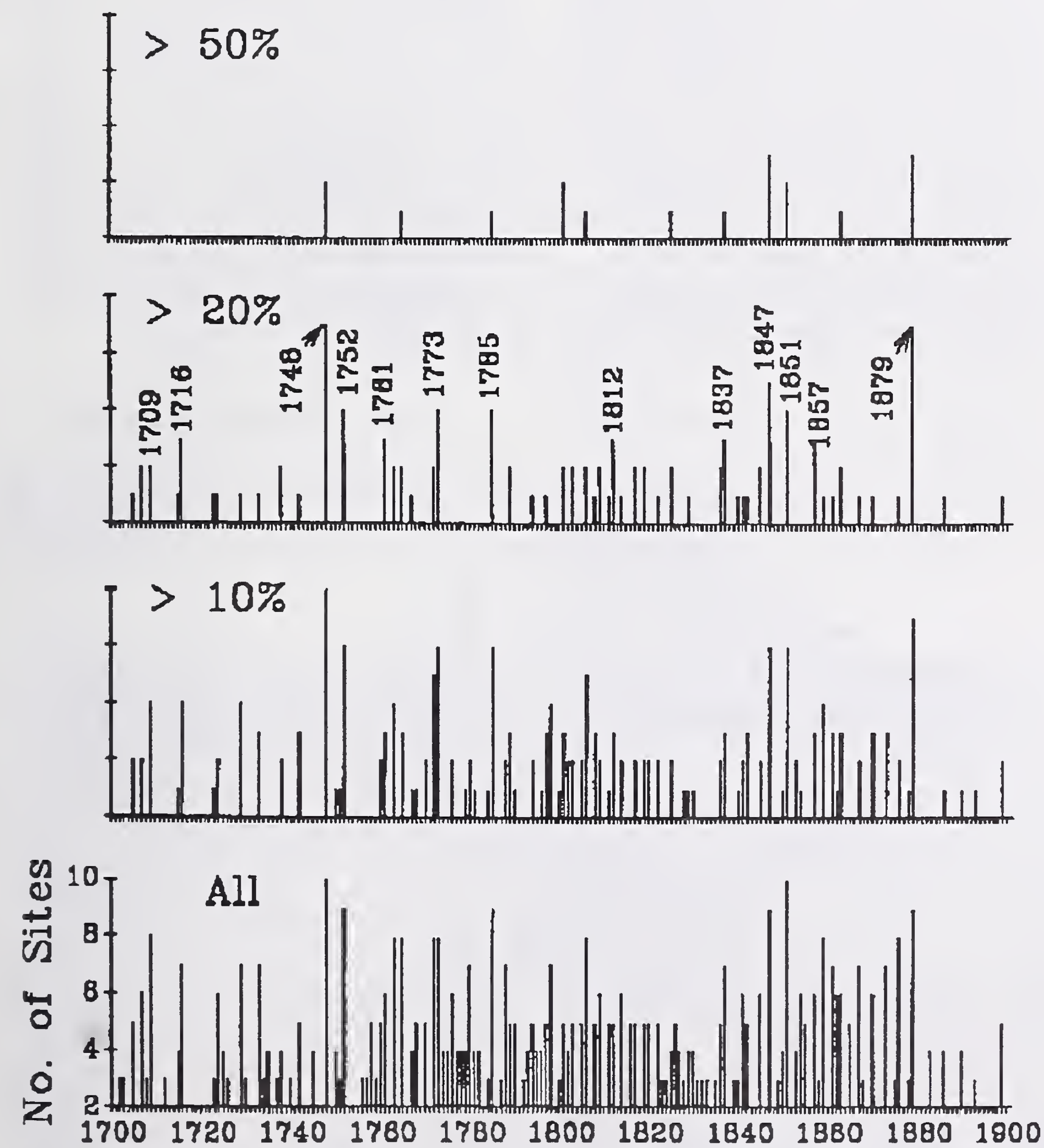

Figure 6.—Fire-area index computed as number of sites (fire-scar chronologies) recording fires per year for the period 1700-1900. Fifteen sites are included. Fires recorded by any tree within the sites are shown in the lowermost plot, while fires recorded by more than 10, 20, and 50% of fire-scar susceptible trees are shown above. Regional fire years are labeled.

Climatic warming is another concern. Regardless of whether or not a warming effect due to increased CO_2 in the atmosphere is already reflected in global weather patterns, there is a building consensus among atmospheric scientists that such an effect, at some unknown intensity, is virtually imminent. Since some of the projected warming scenarios show increasing droughtiness in the southwest (Schlesinger and Mitchell 1985) this effect may also contribute to a worsening fire situation.

From an ecological point of view, the variability in fire regimes is more likely to be important to plant communities than mean values computed for some arbitrary period. For example, unusual long periods without fire may lead to increased establishment of certain species that are intolerant of fire during the first years of life. The co-occurrence of such fire-free periods and wetter climatic conditions may also be extremely important to species with episodic regeneration patterns, such as ponderosa pine. Thus, while statistical summaries of fire chronologies are useful for general comparisons of different forests, fire's influence on the ecosystem is strongly a historical process. Southwestern forests may be more a product of relatively short-term and unusual periods of climate and fire frequency then average or cumulative measures of these long-term histories.

ENSO and Fire

The ENSO-fire teleconnection in the Southwestern Region is clearly inverse, i.e., significantly wetter springs and summers during ENSO events results in a very reduced fire load. Additional work has also shown that the opposite pattern of El Niño, sometimes referred to as La Niña (Kerr 1988), seems to often correspond to peak fire occurrence years (Swetnam and Betancourt, in prep.). The most promising application of

these findings would be a predictive tool for anticipating fire season severity. Work by Bradley et al. (1987) and Andrade and Sellers (1988) has shown that temperature and precipitation response in North American records usually lags 3 to 6 months behind the onset of ENSO events as measured by sea–surface temperatures and the southern oscillation. Thus, the ability of meteorologists to predict ENSO conditions during the winter months before a fire season is likely to improve in coming years (Barnett et al. 1988).

What are some of the management implications of such predictive power? One implication would be for the planning and implementation of prescribed burning plans. Prescribed burning during the typical arid spring and foresummer in the Southwest carries an unknown level of risk due to the uncertain timing of arrival of the monsoon pattern later in the summer, and the possibility of extended dry and windy conditions in the interim. Developing ENSO conditions during winter months, especially those suggesting extreme events, may call for stepping up prescribed burning activities in the spring and early summer because the likelihood of drying conditions later in the season would be lower. Likewise, if further research shows a strong link between La Niña conditions and high fire occurrence, it would be advisable to curtail spring and summer burning during such years. Even if the ENSO phenomena does not prove to be a useful predictive tool for southwestern fire managers, the high likelihood that regional-fire years will recur in the southwest strongly indicates that consideration be given to drying trends during previous and current seasons.

Conclusions

Fire occurrence records for pre-1900 periods document the ubiquity of fire in the southwestern landscape for several centuries prior to the beginning of active forest land management and fire suppression. Through time the dynamic effects of surface fires were second in importance only to the change of seasons. Trees that germinated, established and grew through the majority of their lifespan in a fire regime of repeated surface fires still comprise the vast majority of harvestable timber products in the region, as well as contributing to other forest values. From this perspective of the past, managers must assess the ecological consequences of 80 years of fire suppression in forest types that have adapted to fire over many thousands of years. What will our forests look like 100 or 200 years from now if fire does not play the role it did for centuries before our intervention? The fire histories documented by fire-scar studies remain one of the strongest scientific arguments for incorporating fire into the management of southwestern forests. In making management decisions, ecological knowledge of factors important to the healthy functioning of forest ecosystems must also be weighed against practical considerations, such as the generation of smoke and the hazard of escaped fire.

The synchronous occurrence of large fires in many of the widely scattered forests of the Southwest in particular years is an outstanding feature of both pre– and post-1900 fire records. Severe and very severe ENSO events appear to be consistently associated with reduced fire occurrence in the region. Regardless of whether the ENSO-fire teleconnection proves useful for predictive purposes, this finding and the observation of regional-scale fire years argues that, to a large degree, regional climate patterns control year-to-year fire occurrence. A challenge to researchers and managers is to recognize these patterns and to act in time. Recommended actions include continued and increased effort to reduce accumulated fuel loadings through careful use of fire, and consideration of climatic patterns in fire management planning.

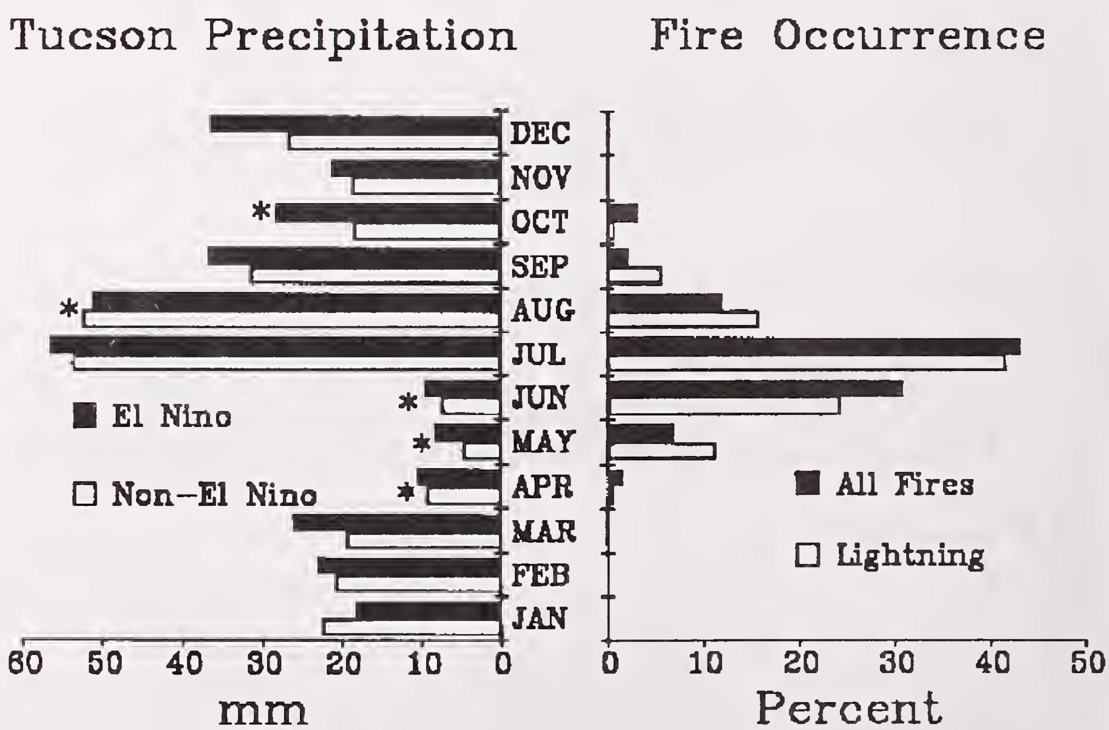

Figure 7.—Distribution of fires by month (1940-1975) in comparison to a precipitation record from Tucson, Arizona (1868-1986). Peak fire activity is during the driest period. Especially critical are the two to three weeks just prior to the "Arizona Monsoon" at the end of June and beginning of July. Wetter months during El Niño years in the winter, spring and foresummer (significant differences indicated with an asterisk, Mann-Whitney test, $p < 0.05$) may explain most of the reduction in area burned per year in the Southwestern Region.

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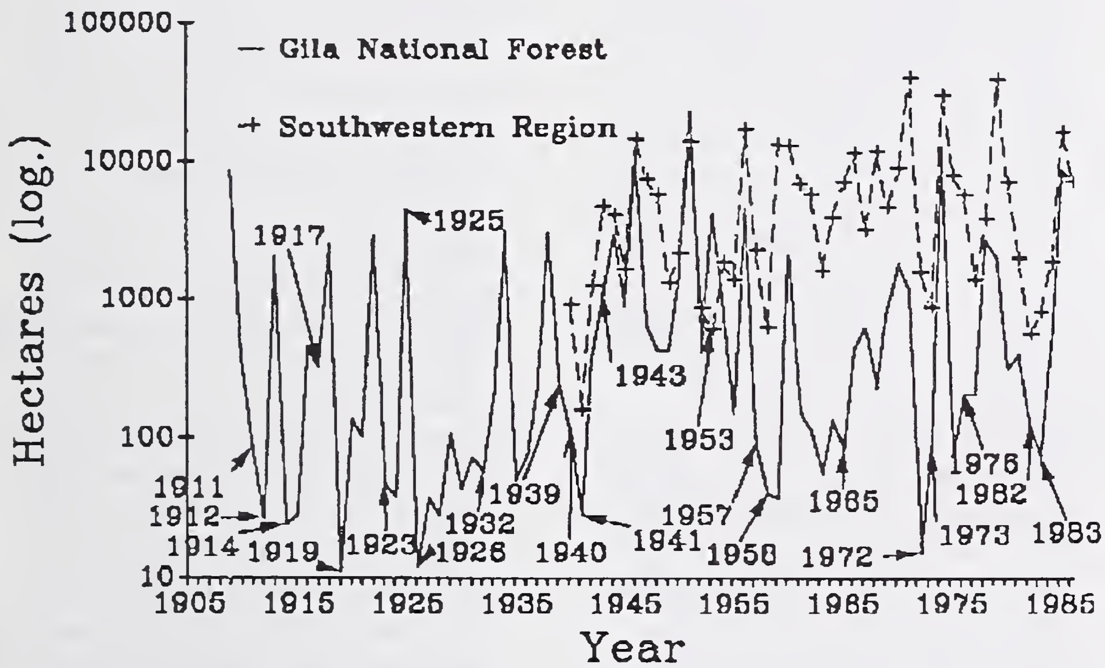

Figure 8.—Area burned per year (logarithmic scale) in Southwestern Region and Gila National Forest in relation to moderate and severe El Niño events. Severe or very severe events according to Quinn et al. (1987) occurred in 1911-1912, 1914, 1917, 1925-1926, 1932, 1940-1941, 1957-1958, 1972-1973, 1982- 1983. All other labeled dates were moderate El Niño

Table 4.—Mean area (hectares) burned during non-El Niño and El Niño events. "n" is number of events included in the grouping and "p" is the two-tailed probability level for Mann-Whitney U test for differences between the means of non-El Niño and El Niño groupings. Intensity of El Niño events was clasified by Quinn et al. (1987) on a scale of weak-moderate, moderate, severe, and very severe corresponding to the groupings below (WM,M,M+,S,S+,VS).

Areas	Non-El Niño years	El Niño years S/S+/VS	M/M+/S/S+/VS	WM/M/M+/S/S+/VS
Gila N.F.	1,582	387	355	1,412
(1909-1986)	n = 52	n = 14	n = 16	n = 26
	p < 0.001	p < 0.001	p = 0.002	
Southwestern Reg.	8,676	957	2,114	3,013
(1940-1986)	n = 34	n = 8	n = 12	n = 13
	p < 0.001	p = 0.001	p = 0.003	

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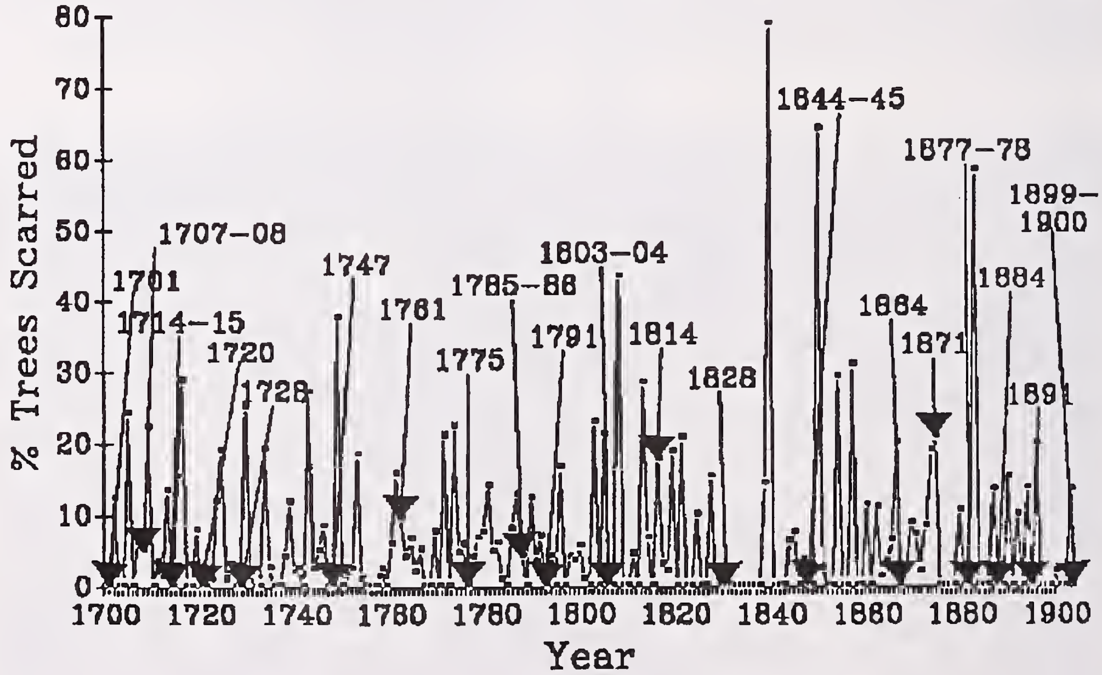

Figure 9.—Severe and very severe El Niño events in relation to percent fire scar-susceptible trees scarred per year from five Gila National Forest chronologies (series 7, 8, 9, 10, and 11 in fig. 1 and table 1). The means for non-El Niño and El Niño years were 6.9% and 3.6% respectively (p = 0.07, Mann-Whitney test).

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Fire Effects on Vegetation and Succession[1]

Malcolm J. Zwolinski[2]

Abstract.—Fire adaptive traits which enhance survival or reproduction of plants are critical in determining post-fire succession. Classical succession concepts which advocate orderly community replacements and retrogression following disturbances fail to describe succession in fire-prone ecosystems. Species attributes and multiple-pathway models provide better post-fire predictors for management.

Although land managers today are expected to predict the short- and long-term ecological effects of different management alternatives, predictions of fire effects on vegetation are particularly difficult due to the wide variability in fire characteristics and vegetation complexes. There is definitely a need for better and more reliable predictions on how various management options affect plant communities and other ecosystem components. These predictions must be incorporated into fire management decision-making and policies (Kessell 1981).

Fortunately, the situation is improving. The past decade has provided a number of significant advances in our knowledge of and ability to predict fire effects and post-fire succession. A better understanding of adaptive characteristics of individual plant species and basic successional processes, coupled with the continued developments in fire behavior models and geographic information system data bases, is already having an impact on fire management decisions.

The purpose of this paper is to briefly summarize the general effects of fire on vegetation, including heat effects and adaptive characteristics, and to examine the past and present concepts of ecological succession following disturbance by fire. The effects of fire on specific southwestern vegetation types and resource values will not be treated here but will be covered in considerable detail by later authors.

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[2]*Professor of Watershed Management, University of Arizona, Tucson, Arizona 85721*

Effects of Fire on Vegetation

Fires in natural ecosystems consume vegetative material, produce residual mineral products, and raise temperatures for short periods. In many fires, plant response depends more on the direct and indirect effects of higher temperatures than on available fuel or release of products. Some heat effects may be immediate and easily observed, i.e., the killing of plant tissue; other effects may be delayed and more difficult to detect, such as damage leading to increased insect and disease susceptibility. This variability can cause problems for managers faced with the task of predicting changes in plant succession and ecosystem dynamics.

Any woody or herbaceous plant can be killed by a fire of sufficient duration and intensity. It is with fires of lower intensity or shorter duration, where only a part of the plant community is killed, that the inherent ability of individual plants to withstand or resist fire becomes significant.

A plant's ability to withstand fire and subsequent heat effects depends upon its heat tolerance and its fire resistance. Heat tolerance is the ability of plant organs and tissues to withstand elevated temperatures, whereas fire resistance is the ability of a plant to survive the passage of a fire.

Heat Tolerance

A plant is directly injured or killed when the temperature of internal living cells is raised to a lethal level. Precise information on the temperatures necessary to kill living plant tissues is lacking (Brown and Davis 1973). Baker (1950) has stated that the heat tolerance of plant protoplasm appears to be the same for all species. This suggests that, although protoplasm coagulation defines lethality for the individual cell, the differences in internal heat effects is due largely to differences in insulation from heat sources.

A temperature of 60°C (140°F) for one minute is often given as the lethal temperature required to kill plant tissue. However, Wright (1970) points out that the temperature at which death of plant tissue occurs is largely dependent upon the tissue moisture content, where tissues with higher moistures are killed at lower temperatures and shorter time intervals. It is possible that a steaming effect may be responsible for increased

heat susceptibility in plants with higher moisture contents.

Not only is heat tolerance dependent on tissue moisture content, but it also varies for different plant parts. For example, seeds are known to be very tolerant of heat (Daubenmire 1968). Chaparral species with hard, thick seed coats are able to tolerate temperatures of 125° to 150°C (260° to 300°F) (Sampson 1944). Wright and Bailey (1982) indicate that seeds covered slightly with soil can be insulated from the effects of a moderate to intense fire. Furthermore, they speculate that grass fires would probably have little or no effect on the mortality of dormant seeds, even if the seeds were lying on the soil surface. Roots, stems, and foliage also exhibit variations in heat tolerance, as can the presence of salts, sugars, pectins, and other plant tissue substances.

Fire Resistance

A plant's ability to survive or recover from the passage of a fire, defined as fire resistance, is dependent upon its food reserve levels and the presence or absence of fire adaptive traits, i.e., protection of growing areas, resprouting, germination of dormant seeds, etc. Plants are more susceptible to damage from fire during flowering and seed formation or during periods of active growth when carbohydrate reserves are low. Since seasonal trends in food reserves vary considerably between plant species, burning at any particular season can be more detrimental for some species than for others.

Fire Adaptive Traits

In fire ecology literature, four fire adaptive traits of plants have been given special attention. These traits are described as those which enhance survival during a fire, such as (1) bark thickness and protected buds which result in vegetative sprouting, and those which enhance reproduction, such as (2) fire-stimulated flowering, (3) seed storage on the plant and fire-stimulated dehiscence, and (4) seed storage in the soil and fire-stimulated germination (Gill 1981).

Bark thickness and protected buds.—The insulating qualities of bark are well recognized. Reifsnyder and others (1967) have stated that the primary factor in determining whether a tree is fire resistant or not appears to be bark thickness. Wright and Bailey (1982) report that trees suffer very little heat damage if the bark thickness is 1.0 to 1.3 cm. Many plant protective mechanisms vary during growth and development so that susceptibility to fire damage may also change with age. In general, bark tends to be quite thin in small trees, considerably thicker at maturity, and declines in thickness with senescence.

Plant survival following fire can frequently be attributed to the location of meristems and the protection buds receive from reaching lethal temperatures. Packham (1970) noted that the amount of rising convective heat from a surface fire is three times that of radiated heat. The exposed location of terminal and lateral apical buds of many shrubs make them highly susceptible to top-killing from rising convective heat, whereas surface and subterranean buds are well-protected against major heat inputs. The basal meristems of a number of grass and forb species provide a distinct fire survival advantage. Basal sprouting may be common in shrubs after fire has destroyed the foliage. Sprouting is thought to be an ancient adaptation which occurs when the foliage is removed by an external agent (Chandler and others 1983). While the leaves are alive, an inhibiting factor prevents bud activity, but when the foliage is killed the inhibition disappears and the dormant buds begin sprouting. Resprouting after fire appears to be related to the age of the plant, stem size, season of the year, fire frequency, and fire intensity.

Fire-stimulated flowering.—Among fire-resistant plants, the phenomenon of fire-stimulated flowering has been observed (Chandler and others 1983, Gill 1981). Although not widely reported in the western United States, a number of observations have been made in Australia, Israel, New Zealand, and South Africa. This trait appears to be most frequently associated with the monocotyledons, especially the amaryllis, grass, lily, and orchid families.

The mechanisms responsible for the triggering of flowering by fire are still unknown. The results, however, are that prolific flowering is followed by increased seedling establishment. This may be a consequence of greater seed set, lower on-plant seed predation, or lower predation of seeds following dispersal (Gill 1981). Chandler and others (1983) speculate that either a change in diurnal temperatures following fire or an increase in the amount of light reaching the soil may cause heating of the floral reproductive organs and induce flowering. The removal of shrubs could also promote more effective pollination of certain species by insects and wind.

Seed retention and fire-stimulated dehiscence.—For many shrubs and trees, retention of seeds on the plant and stimulation of dispersal is an important mechanism for survival in fire-prone environments. Two examples of tree species that possess this trait are jack pine *(Pinus banksiana)* and lodgepole pine *(Pinus contorta)*. A number of other pine species have cones that dehisce (split open) after fire or under the effects of fire, a trait known as serotiny. The cone scales of these pines are held closed by a resinous or waxy material sensitive to high temperatures. As fire passes and heat melts the resins, the cone scales exfoliate and seeds are released. Lodgepole pine is particularly interesting because its cones can vary from serotinous to freely dehiscent (Brown 1975; Lotan

1975, 1976). In stands where fires are frequent, serotinous cones are common, whereas cones are freely dehiscent in forests where fire is less frequent. Lotan (1975) also reported that young lodgepole pine trees tend to have open cones while older trees possessed either open or closed cones.

When seeds from serotinous cones are released by fire, they often fall on a favorable seedbed. The ash and minerals provide nutrients while the lack of overstory foliage increases the amount of sunlight reaching the soil to assist in seedling germination and growth.

Seed storage in the soil and fire-stimulated germination.—The storage of seeds in the soil and stimulation of their germination by fire can be seen as an adaptive trait. Seed germination in recently burned plant communities can usually be attributed to the release of seeds retained by the plants or to the germination of seeds stored in the soil from past years. Some transport of seeds to the burned area from sources in adjacent unburned sites is also possible.

Hardseededness is a term applied to seeds with a physical barrier to germination, commonly typified by the lack of imbibition, swelling, and softening when exposed to water (Gill 1981). However, when the seed coat is scarified, germination does occur. A number of woody and herbaceous species, perennials, and annuals exhibit this trait. Fire is one of the mechanisms capable of scarifying seeds and breaking seed dormancy. Similar results can be achieved by the movement of seeds in ephemeral streambeds and softening of seed after passage through birds and animals (Gill 1981).

Research by Muller and others (1968) on the allelopathic conditions in California chaparral indicates that profuse flowering and appearance of herbaceous species after fires is due to the destruction of a chemical inhibitor produced by the woody plant cover and deposited on buried seeds. The authors concluded that heat is not required for germination, but germination is stimulated by the removal of the inhibiting ligneous extracts. The degree of seed coat inhibition removed is dependent upon the intensity and duration of heat exposure. When woody plants return, the inhibition becomes reestablished; however, the seeds remain viable until another fire takes place.

Gill (1981) refers to these four vegetation adaptations as the "classical" fire traits. He points out that there are other developmental patterns of plant species which could be considered fire adaptations. Mentioned as possibilities are seed burial, plant longevity, chemical composition, time to first flowering, and patterns of leaf shedding. Many adaptive mechanisms are able to facilitate the reproduction or survival of plant species in fire-prone environments.

Fire Characteristics

Chandler and others (1983) state that adaptive traits have been considered in relation to the occurrence of a single fire, but, in actuality, an individual plant may be exposed to several fires, each with different fire characteristics and fire effects. To determine the significance of adaptive traits consideration must also be given to the life cycle of the species and fire regimes to which the species is subjected.

Fire frequency determines the vegetative composition of an area by selecting species which will continue to occupy a site. A species can be removed if fire occurs too often, too early, or too late in its life cycle. For instance, a non-sprouting species may be lost if fire occurs before seed has been produced, or if fire occurs after the species has died and the seed pool is unavailable (Chandler and others 1983). Two strategies typically characterize the response of different species to fire frequencies — those that sprout can withstand repeated fires while those that produce seed are favored by infrequent fires (Keeley 1981).

Understanding the effect of fire intervals is important because it can significantly affect the survival probability of an individual, population, or species. In general, a pattern of less frequent fires burning with higher surface temperatures due to increased fuel accumulations, and vice versa, has been recognized. Fire regimes, then, modify the evolution of plants and any changes in these fire regimes will precipitate a change in the floristic composition and structure of a vegetative type.

The season of fire occurrence is a very important factor affecting plant survival and flowering. Frequently, substantial fire effects differences can be observed between spring and fall burns. Spring fires tend to damage annual grasses that emerge following winter rains and have not had the opportunity to produce seed. On the other hand, many perennial species are still dormant during a spring fire and could resprout later in the season (Wright 1969, 1974).

The size of the area burned by a stand-consuming fire can influence recolonization if plants are unable to regenerate by sprouting. Seed carriers, such as wind or animals, may not provide adequate distribution of seeds if the burned area is extensive.

Vegetative responses to intensity and duration of heating vary considerably depending on the natural role of fire in an ecosystem. With fire-sensitive species, a low-intensity fire may be very damaging, while a moderate- to high-intensity fire in a vegetative type dependent upon fire may stimulate reproduction and cause little change in floristic composition. Fire intensity is one of the more difficult fire characteristics to assess. Some observers rely on the visual post-fire changes evident on vegetation or soil, others install temperature sensors or attempt to correlate intensity with flame lengths using different indices.

A fire that creates high surface temperatures for a long duration can result in heat penetrating into the surface soil. The survival of subterranean organs, i.e., roots, rhizomes, and seeds, is dependent on the depth of heat penetration. If penetration is extensive, organs are killed and reestablishment of the species will be difficult. A low intensity fire does not destroy these organs which allows the vegetation to become reestablished on the site quickly.

Effects of Fire on Succession

The preceding discussion was specifically directed to heat effects, fire adaptive traits, and fire characteristics. This information can now lead to an examination of ecological succession in a fire-prone environment. Before considering the present-day concepts of fire-disturbed succession, it is important to look back at the early or classical version of succession and to see why it fails in disturbed ecosystems.

Classical Succession

The classical definition of succession is simply the replacement of the biological community of an area by another community. Classical Clementsian succession can be described as (1) an orderly process of community change which is reasonably directional and predictable, (2) resulting from the modification of the physical environment by the present community which creates conditions suitable for the establishment of another community, and (3) ultimately reaching a biologically stable ecosystem stage (climax community) (Clements 1916, 1936).

Clements has also identified the successional development of vegetation as a sequence of five processes: immigration, establishment, site modification, competition, and ecosystem stabilization.

Two types of succession are often recognized — primary and secondary. Primary succession is the development of communities on newly exposed bare areas which have not previously supported vegetation. Primary succession, also known as autogenic or self-induced succession, starts with a pioneer stage (usually mosses and lichens) and progresses to larger and taller species, each creating a microclimate or soil condition which induces the emergence of a new community and the loss of the old.

Secondary succession is the sequence of vegetation development on areas which have supported vegetation but now vegetation has been destroyed, in part or in total, by an agent such as fire, flood, windstorm, etc. The term allogenic (externally-induced) is often used with secondary succession because changes are precipitated by forces independent of the community itself. Secondary succession can appear at any stage of primary succession and, theoretically, causes a retrogression or resetting of succession back to an earlier stage. The essential distinction between primary and secondary succession is that pioneer communities of a secondary succession receive the benefit of soil already in place.

Probably one of the most confusing aspects of classical ecological succession is the concept of a climax community. A postulate advanced by Clements (1936) and his followers states that community succession leads to a climax and that the concept of succession cannot be separated from that of climax. Clements felt that with sufficient time and competition, an undisturbed plant community would approach the same species composition and structure in a given climatic region. This monoclimax or climatic climax concept placed only secondary importance on other site factors such as soil, topography, or repeated disturbances. Over time, ecologists recognized that stable vegetative communities can also be attributable to site or other environmental conditions. This has led to a number of additional definitions of climax attributes such as polyclimax, subclimax, physiographic climax, edaphic climax, disclimax, preclimax, and postclimax.

Notwithstanding the debate on climax, ecologists acknowledge that some ecosystem changes accompany the successional growth and maturation of vegetational stages. For example, there appears to be a progressive increase in community complexity and diversity from early to mature stages, an increase in the total biomass and gross productivity, and an increasing development and maturation of soils.

Fire Disturbances

The classical concept of succession subscribes that, following a disturbance, such as fire, the present community disappears and is replaced by an earlier vegetation type which results in a retrogression or backing up of succession. Succession then moves forward through the intermediate or seral stages to the stable, climax community. This classical succession has been described in the literature as the facilitation model (because of its emphasis on changes within successional areas facilitating the establishment of new species), the relay floristics model (due to the replacement or relay sequence of successional stages or seres), or the single pathway model where succession follows a predictable sequence of steps (Connell and Slatyer 1977, Egler 1954).

Not all successions necessarily proceed stepwise through to a climax community. If disturbances, such as fire, are part of the natural environment of a plant community, then the term climax loses meaning since all species that persist are climax species. Climax implies stability; however, plant communities cannot be completely stable. With different

ages and lifespans, there are weak and overmature individuals that disappear and are replaced. Openings, either natural or human-caused, allow the establishment of new individuals or species. Thus, communities are characterized by continual change rather than by stability. It is interesting to note that Patterson (1986) has recommended that the word "climax" be stricken from forest terminology. He argues that Clementsian climax theory is known to be based on assumptions that cannot be met, and that continual changes in time and space make ecosystems dynamic instead of stable.

As the failure of classical succession concepts to describe vegetational changes resulting from disturbances became more apparent, scientists began to propose other models of succession. In 1977, Connell and Slatyer suggested two additional pathways of succession. They identified these as the tolerance and the inhibition models; both were attempts to explain the site occupancy of certain species following disturbances. The tolerance model describes the situation in which later species are successful whether or not earlier species have preceded them. The inhibition model considers the condition where later species cannot grow to maturity in the presence of earlier ones. Although these two models provide some insight on species establishment, they did not deal effectively with species changes being observed following fire disturbances.

As more information on fire disturbances and succession became available, Noble and Slatyer (1981) stated that several generalizations could be made. These were:

1. Species composition immediately following a fire is dependent upon propagules (reproductive structures) which have arrived from adjacent areas, have persisted through the fire, or have resprouted vegetatively from organs surviving the fire. Therefore, it is not surprising that replacement sequences following fire tend to be reproducible and often lead to development of communities similar to those existing before fire (initial floristic composition) or existing on nearby undisturbed areas.

2. Shortly after fire, there is a surge of recruitment and regrowth under conditions of low competition for site resources.

3. There is a slowing in recruitment following the initial surge as individuals become established and are more difficult to displace.

4. Further recruitment of new species may be facilitated by prior occupancy, but it may also be restricted or inhibited by present occupants.

5. In the absence of further fires, species that are long-lived and those that can regenerate and grow under an adult canopy will become dominant.

For understanding successional pathways following fire, evaluation of responses by individual plant species leading to community development may be considerably more useful than the community replacement approach of Clements. Fire-induced succession is related to the adaptations possessed by individual plant species to colonize, grow, and survive. Therefore, specific life-history characteristics of key species in a particular community can be used to describe successional patterns which follow fires of varying intensities and frequencies.

Fire is now recognized as one of the most common disturbances of natural ecosystems and it is becoming widely acknowledged that the properties of individual species are one of the key factors in determining successional patterns in fire-adapted communities. Noble and Slatyer (1977, 1980, 1981) and Noble (1981) have provided the foundation for much of today's work on plant adaptive traits and post-fire succession. These authors sought to identify a small number of attributes which are vital to terrestrial, higher plant community species which occur in areas subject to recurrent fires. They developed a "multiple pathways" model for predicting major shifts in species composition and dominance in fire-prone ecosystems which is based on selected plant "vital attributes."

Three main groups of species attributes which are vital in a vegetation replacement sequence have been recognized. These are (1) **method of persistence** (the method of species arrival after, or persistence during, a fire), (2) **mechanisms for establishment** (attribute associated with the site conditions in which the species become established and grow to maturity), and (3) **critical life stages** (time needed for the species to reach critical periods in its life history, i.e., reproduction, maturity, senescence, extinction).

In recent years increasing numbers of authors have reported the use of the vital attributes approach to describe successional changes following fire (Hobbs and others 1984, Noste and Bushey 1987, Rowe 1983). Some excellent fire ecology information and fire management guidelines using species attributes and fire characteristics for specific forest habitat types in Idaho and Montana have been published (Crane and Fischer 1986, Fischer and Clayton 1983, Kessell and Fischer 1981).

Although it is evident that this method is providing managers with a more realistic prediction of post-fire succession, there are situations where its application is difficult. For example, the wide variability in cone serotiny for lodgepole pine has direct

applicability to its reproductive success after fire. Some difficulties also arise because of the lack of vital attribute information on key plant species. It is recognized that this approach works best for forest and brush overstories, but little attention has been given to understory herbaceous species. As additional data on plant adaptations, life histories, and site-specific fire characteristics become available, the use of the multiple pathway vital attributes model of succession will become a more valuable tool for the land manager in fire prone areas.

Management Implications

The land manager attempting to predict the effects of fire on vegetation and subsequent successional sequences is facing a difficult task. This individual is being asked to integrate information which ranges from plant heat tolerances, fire resistances, and plant adaptations, to specific fire and weather characteristics, into a reasonably comprehensive and meaningful fire management plan. The magnitude of the task is readily apparent. Thus, a major function of this paper was to provide an overview of fire effects and their relationship to succession and to furnish managers with information on changes and new approaches for use in decision-making.

One of the more important points for today's land managers who deal with fire is to recognize that the application of classical succession concepts to predict post-fire vegetation patterns is not appropriate. Therefore, managers should become as familiar as possible with the individual plant species in their area and, particularly, try to determine what attributes these species have to survive or persist following fire. Some of this latter information may be available in the literature but, ultimately, close field observations and records of plant responses during and after fire by researchers and managers together may be most valuable. Researchers and managers need to determine precisely what information is required to do the job (USDA FS 1981). Although some researchers disdain the concept of "good enough," often managerial objectives can be adequately met by procedures and models that are neither the "best" nor the "most complete." The cost of improving or expanding such procedures may not be financially justifiable to the manager or the agency. A program of cooperation and communication between research and management will yield greater benefits than individual efforts.

An example of such cooperative efforts is the development of a Fire Effects Information System (FEIS), a new computer-based storage and retrieval system. This FEIS is being developed at the Intermountain Fire Sciences Laboratory in Missoula by Forest Service prescribed fire and fire effects research personnel and the Computer Science Department at the University of Montana. The Bureau of Land Management and the National Park Service have provided support for the project. The purpose of the system is to provide land managers easy access to up-to-date information on fire effects in plant communities and associated individual plant and animal species. Managers interested in this system may wish to contact the Intermountain Fire Sciences Laboratory, the Boise Interagency Fire Center, BLM State Offices in Idaho, Colorado, Nevada, and Oregon, or personnel at Wind Cave National Park.

In the coming years continued advances will be made in our understanding of and ability to predict post-fire plant succession. Work has already progressed on several fronts, including the knowledge of basic successional processes, the adaptive characteristics of species populations, the effects of fire intensity and periodicity, and the ability to integrate successional sequences with ecologic and managerial concerns.

The challenge of the 1990's will be to continue to build upon this knowledge base and to expand application to include how fire disturbances and their management affect other terrestrial and aquatic community components. Fire impacts on water quality, nutrient cycles, ecosystem dynamics and energetics, fish and wildlife, fuel complexes, and soil properties are directly and indirectly related to floristic changes and successional patterns. Additional information on these components in a fire disturbed community will, undoubtedly, substantially enhance the value and applicability of fire management plans.

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Effects of Fire on Cultural Resources[1]

John Lissoway[2] and Judith Propper[3]

Abstract.—Over the past 10 years, considerable information has been gathered on the effects of fire and fire suppression activities on cultural resources in the Southwest. A review of this information is presented, and recommendations are offered on how damage to cultural resources can be minimized through integration of cultural resource considerations into fire management activities.

Fire managers today are involved in developing programs to accomplish objectives far more complex and difficult than ever before. In meeting these objectives they must also ensure the protection of many very significant resource values. One important challenge and obligation for fire managers, particulary in the Southwest, is to identify, understand, and protect cultural resources. This paper is intended to organize for the fire manager much of the current information on the importance of cultural resources and how fire affects them. Based on this, management considerations and recommendations are offered on how to protect these irreplaceable values.

CULTURAL RESOURCES AND WHY WE PROTECT THEM

In the broadest sense, cultural resources are the surviving traces of past peoples and cultures. Cultural resources include artifacts, such as spear points and pieces of pottery, as well as the remains of pit houses, cliff dwellings and log cabins. Other evidence includes rock art, fire hearths, stone quarries, trash deposits, and remnants of fields or canal systems. All of these remains hold clues to past ways of life. Over time their study will help us better understand the course of human events and perhaps why societies succeed or fail.

Well over 95% of the story of human life in the Southwest took place before the arrival of Europeans. This immense span of time, before the advent of written records, is called prehistory. It encompasses more than ten thousand years of cultural adaptation and change, from the PaleoIndian mammoth hunters of the closing Ice Age, to the beginnings of agriculture and settled life, to the architectural wonders of Chaco Canyon. All we can hope to learn about these prehistoric people and their struggle to survive is contained in the archeological record they left behind.

With the arrival of Coronado in 1540, the historic period begins. Cultural resources from this period document the settlement and development of the Southwest under Spanish, Mexican, and finally American rule. Remains include missions, ranches, forts, homesteads, mines and logging camps. Also included are historic period pueblos and other Native American sites. Cultural resources of the historic period can provide a wealth of information on local history and daily life, information not always recorded in the books and journals of the day.

There is little doubt that the Southwest contains the nation's most spectacular and best preserved record of the past. Many sites, for example fortifications and large pueblo ruins, are easy to recognize. Others, like pit house depressions and artifact scatters are harder to detect. The importance of cultural resources lies not just in the artifacts they contain, but in the relationships between the artifacts and features present at a site, both above and below the ground. It is through the study of these relationships that archeologists can determine the sequence of events at a site and the activities and ways of life represented. In addition to scientific values, cultural resources may hold special social or religious significance for living people, for example Native American ancestral sites and shrines. Finally, cultural resources have educational and interpretive value because they are the tangible core of the nation's cultural heritage. More and more Americans are becoming interested in the past and are seeking opportunities to visit and learn about archeological and historic sites.

Although many cultural resources in the Southwest have survived for hundreds or thousands of years, they can be wiped out in a moment by heavy equipment and construction activities. In that sense, cultural resources are extremely fragile. They

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[2]John Lissoway is Chief of Natural and Cultural Resources Management at Bandelier National Monument, Los Alamos, New Mexico.

[3]Judith Propper is Regional Archeologist for the Southwestern Region, USDA Forest Service, Albuquerque, New Mexico.

are also non-renewable. Once a site is destroyed, the information it contained is lost forever. Each time this happens, our chances of eventually understanding a particular culture or time period are diminished.

Because of concern over the loss of cultural resources due to development activities and looting, Congress has passed a number of laws to protect sites on public lands. The Antiquities Act of 1906 and the much stronger Archaeological Resources Protection Act of 1979 prohibit the unauthorized excavation or removal of archeological resources from Federal lands. The National Historic Preservation Act, passed in 1966, directs Federal agencies to inventory, evaluate, and protect cultural resources under their jurisdiction. It also directs agencies to take into account the effects of their activities on cultural resources in order to ensure that important sites are not inadvertently destroyed. Today, consideration of cultural resources, including field surveys when information is inadequate, has become a routine part of the project planning process in agencies like the Forest Service, Park Service, and the Bureau of Land Management. Steps have also been taken to begin integrating cultural resource considerations into fire management activities.

EFFECTS OF FIRE AND FIRE MANAGEMENT ACTIVITIES ON CULTURAL RESOURCES

Only within the past 10 years or so has serious consideration been given to assessing the effects of fire on cultural resources in the Southwest. Most of our current information is derived from observations and studies carried out in the aftermath of wildfires. Although plans have been developed to test the effects of fire on various types of cultural remains (Kelly and Mayberry 1979; Jones and Euler 1986), little has been published on the results of such experiments.

The fires in the Southwest for which cultural resource information is available include the following:

1. **Little Moccasin Canyon Fire, Mesa Verde National Park, 1972.** This fire burned 1,080 acres within the Park. Pinyon, juniper, and oak brush were the primary fuels. Following the fire, a cursory inspection of several sites revealed considerable damage to masonry pueblos and surface ceramics (Switzer 1974).

2. **Dutton Point Fire, Grand Canyon National Park, 1976.** This fire burned 321 acres. The vegetation type was primarily ponderosa pine and Gambel oak with a sage understory. An intensive survey following the fire documented effects on eight small masonry sites (Jones and Euler 1986).

3. **La Mesa Fire, Bandelier National Monument, 1977.** This fire burned over 15,000 acres within Bandelier National Monument, the Santa Fe National Forest, and Department of Energy lands. The area is characterized by long finger-like mesas and steep canyons, with ponderosa pine and pinyon-juniper as the dominant vegetation types. This was the first fire in which archeologists were called in to help crews avoid cultural resources during fire line construction. Following the fire a survey was conducted of all hand lines, dozer lines, heliports, and other areas of disturbance (Traylor et al. 1979). One hundred sites, mostly small, masonry pueblos, were recorded; of these, 58 were burned to some degree. Test excavations were also conducted at four sites to assess the effects of the fire on subsurface materials. This is the most comprehensive study available on the effects of fire on cultural resources.

4. **Radio Fire, Coconino National Forest, 1977.** A total of 4,600 acres were burned on the slopes and summit of Mt. Elden. A variety of vegetation types were involved including ponderosa pine, ponderosa pine-chaparral, and pinyon-juniper. Following the fire, archeologists surveyed the entire burn area. Sixty-five sites were recorded, primarily small masonry pueblos (Pilles 1984). In addition, four sites were tested.

5. **Jacket Fire, Coconino National Forest, 1977.** Approximately 290 acres of pinyon-juniper woodland were burned. A portion of the burned area was surveyed, and eight small masonry pueblos and four artifact scatters were recorded (Pilles 1984).

6. **Wallace Fire, Coconino National Forest, 1979.** A total of 645 acres of dense ponderosa thickets were burned. Subsequent survey identified 10 small masonry pueblos, 2 pit houses sites, and two artifact scatters (Pilles 1984).

Two major types of damage to cultural resources have been identified in these studies: damage caused by the direct and indirect effects of fire itself; and damage caused by fire suppression and related activities.

Effects of Fire on Cultural Resources

Anywhere from 67% to 90% of the cultural resources within the above

fires were burned to some extent. Three major factors are thought to be involved in determining the nature and extent of fire damage to cultural resources (Traylor et al. 1979:130): 1) fire intensity; 2) duration of heat; and 3) heat penetration into the soil. While many variables, including wind, humidity, and topography, influence a fire's intensity, fuel load is thought to be the most important determinant. Due to years of fire suppression, fuel loads in many areas of the Southwest are probably much heavier than ever before. Likewise, the potential threat to cultural resources may be far greater than in the past.

In the La Mesa Fire, the severity of burning at a site seemed to correspond closely to the density of the fuel load adjacent to and on the site (Traylor et al. 1979:131,134). Although temperatures were not recorded, it was estimated that temperatures of 100-300 degrees Celsius probably characterized lightly-burned grassy areas. Temperatures in the 300-500° range were estimated for fires in the pinyon-juniper vegetation type, and temperatures of 800°C or more may have been reached in ponderosa pine stands.

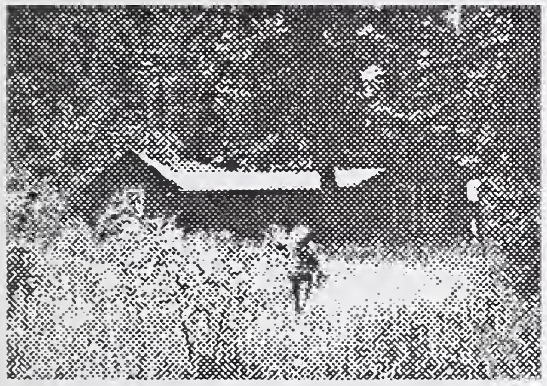

Figure 1—Wooden structures, like this administrative site no longer in use, are highly susceptible to destruction by wildfire.

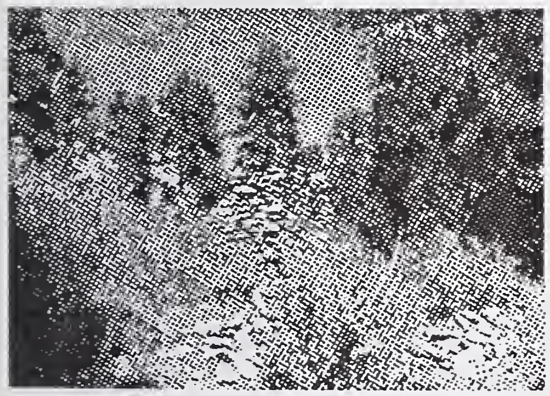

Figure 2—Prehistoric ruins, including masonry architecture, can be damaged by fire as well as by fire suppression activities.

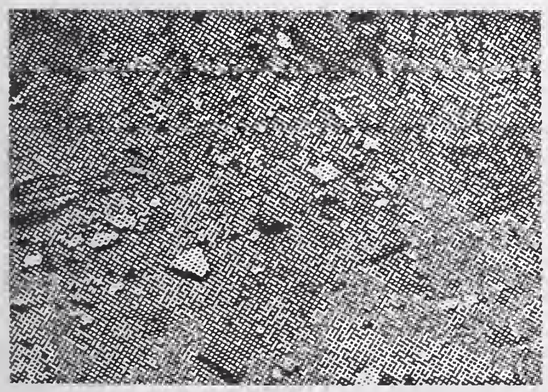

Figure 3—Artifacts on the surface, like this ceramic sherd scatter, can be damaged by fire if temperatures are hot enough.

Direct Effects

The following summarizes what we now know about how fires directly affect various types of cultural materials. Because temperature and pre-fire data are not available for the fires discussed above, comparisons and generalizations must remain fairly limited.

Wood.—Obviously, wooden structures such as frame building and log cabins (fig. 1) can be destroyed by fire. Exposed wooden beams in pueblo ruins are also susceptible to damage or destruction. In both the La Mesa and Radio Fires, remains of historic buildings were destroyed.

Building stones and grinding stones.—Stone masonry (fig. 2) can be damaged by fire. Discoloration, severe cracking, spalling, and disintegration of sandstone building stones occurred in the Little Moccasin Canyon Fire at Mesa Verde. The same was true of tuff masonry in the La Mesa Fire at Bandelier. Effects undoubtably vary with the condition and type of material. On the Jacket Fire, sandstone grinding stones were frequently cracked, whereas those made of basalt were only fire blackened (Pilles:1984:11). Subsurface masonry in the La Mesa Fire was generally unaffected.

Lithic artifacts and raw materials.—While breaking and spalling of chert and obsidian artifacts can occur, effects have generally been limited to discoloration and development of a dull, dark patina. In areas of intense heat on the Radio Fire, some obsidian artifacts were partially melted. In areas where obsidian sources are present, thermal fracturing can obscure evidence of prehistoric quarrying activity (Pilles 1984:11).

Ceramics.—Ceramic remains (fig. 3) can be either carbonized or oxidized by fire resulting in smoke discoloration and changes in paste characteristics. In the La Mesa Fire, surface designs were burned off or obliterated in some cases, making identification impossible. In the Little Moccasin Canyon Fire, spalling and separation along coil lines were observed. Presence of a shiny black residue, possibly pine pitch, has been noted in several fires.

Bone, plant remains.—Organic materials, like bone, seeds, fibers, etc., can be charred or destroyed, even at fairly low temperatures.

Pollen.—Surface pollen can be destroyed at temperatures above 300° C (Traylor et al. 1979:155).

Samples for dating.—Analysis from test excavations following the La Mesa Fire indicated that fire can affect thermoluminescence and obsidian hydration dates obtained from surface artifacts. Results of carbon–14 tests were inconclusive. If wooden beams are destroyed, opportunities for obtaining tree ring dates will be lost.

Rock art.—Although no rock art sites were present in the fires studied thus far, petroglyphs and pictographs would be especially vulnerable to damage by scorching or spalling.

Indirect Effects

Cultural resources also can be damaged by the indirect effects of fire. Factors that have been identified include:

Erosion.—Erosion often occurs in the aftermath of fires due to destruction of vegetation cover and loss of organic material in the soil. On the Radio Fire, run-off following the fire cut gullies up to 10 feet deep in some places (Pilles 1984:11).

Falling trees.—Trees killed by fire eventually are uprooted. This may result in structural damage and artifact displacement.

In summary, available information indicates that wood and other organic materials are most susceptible to destruction by fire, although other remains on the surface may be severly damaged. The general opinion is that relatively cool fires, below 300 degrees Celsius, probably will not cause permanent damage to most inorganic materials. Also, excavation data currently indicate that buried cultural materials in deposits 5 centimeters or more below the surface will not be harmed by most fires (Traylor et al. 1979:148, Pilles 1984:12). One exception would be burning roots, where temperatures as high as 1500 degrees Celsius may occur (Traylor et al. 1979:132)

Effects of Fire Management Activities on Cultural Resources

In most of the fires discussed above, the most severe impacts to cultural resources occurred, not as a result of the fire itself, but as a result of fire suppression and rehabilitation activities. Three major types of damage have been identified (Traylor et al. 1979:111): 1) destruction of architecture and associated information; 2) displacement of artifacts; and 3) destruction of artifacts. Much of the following information is derived from observations on the La Mesa Fire and other fires in the Southwest.

Direct Effects

Fire line and helispot construction.—Fire suppression activities that involve use of heavy equipment pose the greatest threat to cultural resources. Dozer blades and tracks can cause severe damage in and around sites by cutting deep into soils, destroying architecture, and displacing surface materials. On the La Mesa Fire, only two sites were impacted by initial dozer line construction which was monitored by archeologists; however, when these lines were later widened, 15 sites previously avoided were severly damaged. Ten sites were damaged by dozer line construction on the Radio Fire. Hand line and helispot construction, especially at night, can result in exposure of subsurface deposits and may potentially affect structural sites by loading them with discarded vegetation.

Fire retardants.—Some damage to exposed walls resulting from retardant drops by air tankers and helicopters has been observed. Corrosive effects on cultural materials have yet to be analyzed.

Mop–up activities.—During mop-up, damage may result from any of the following activities: engines and other vehicles moving around inside the burn; moving hose; digging roots and stumps in or around a site: and snag felling operations.

Rehabilitation activities.—Activities associated with fire rehabilitation such as water bar construction and installation, berm leveling, equipment used for reseeding, planting, salvage logging and fuelwood collection could have damaging consequences to sites and materials.

Indirect Effects

Surface collecting by fire personnel has occurred on a number of fires; however, once informed of the importance of leaving artifacts in place, crews have been very cooperative. Wildfires can present management with follow-up resource protection problems due to the effects of exposing previously unknown or inaccessible cultural sites and materials to theft or vandalism.

Prescribed Fires

In the case of prescribed fire activities, many similar effects can occur. Managers must be aware that burn plot preparation work such as line construction, porta-tank and hoselay setup, snag felling, pre-treatment of fuels with chemicals, mechanical fuel reduction, vehicle and/ or packstock movement in and around the area, and mop–up and patrol operations can affect cultural sites or materials.

MANAGEMENT IMPLICATIONS AND RECOMMENDATIONS

Training on the nature and protection of cultural resources has been included in interagency fire management training programs for a number of years, and general awareness of the need to consider cultural resource values has increased significantly. Integration of cultural resource considerations into fire management policies and plans probably varies by agency. National Park Service fire policies (NPS-18) address the need to reintroduce fire back into many largely fire-dependent ecosystems, consistent with the protection of the religious rights of Native Americans where applicable, and cultural sites with associated materials. Especially in Southwestern parks, fire management plans must contain guidelines and constraints on fire suppression and fire use, along with procedures to conduct post-fire salvage of artifacts as part of rehabilitation.

In the Southwestern Region of the Forest Service, cultural resource considerations are addressed in some,

but not all, National Forest fire management action plans. Archeologists or other personnel with cultural resources training are usually involved in monitoring use of heavy equipment on fires. Cultural resources are given routine consideration in prescribed fire planning and in rehabilitation activities.

The following recommendations are intended to provide the manager with a checklist of considerations in developing and/or improving their overall program.

Planning and Preparation

1. Obtain baseline cultural resource inventory information, the product being a thematic map or overlay. (Note: Site location information is protected under the Archaeological Resources Protection Act; therefore, managers need to provide for security of this data.)

2. Consult with the archeologist or historian on the relative significance of cultural resources involved (for example, National Register status or potential).

3. Identify and map cultural values at risk in consultation with the archeologist or historian.

4. List how fire affects the cultural materials listed.

5. Consider ways to reduce the risks to highly vulnerable sites by reducing fuel loads, constructing fire breaks, etc.

6. Outline any Native American concerns in consultation (to determine times, seasons to restrict burning, etc.).

7. Identify and present training in cultural site identification, protection measures, etc. to fire personnel.

8. Train and fire-qualify local archeologist(s) to line locator, dozer supervisor, resource advisor, or other jobs as needed by the manager.

9. Identify and classify fire management zones or units based on objectives, fuel type, and values at risk.

10. In consultation with archeologists, prepare a section on cultural resource protection steps in the area fire management plan.

11. Prepare or update the Line Officer's Briefing Statement with a section on specific constraints for the incoming Incident Commander. Preassign a qualified archeologist to present this section to the incident management team briefing.

Minimizing Impacts of Wildfires

1. Include consideration of cultural resources in determining appropriate supression response, i.e. confinement, containment, or control strategy.

2. Consider restricting the use of heavy equipment in culturally sensitive areas.

3. If heavy equipment is used, assign one or more fire-qualified archeologists to flag sites and other cultural features during suppression through mop–up and rehabilitation. (Identify a standard color and method for site marking and assure that all personnel know how cultural resources will be marked.)

4. Make available a cultural resource specialist to assist with shift briefings for overhead, especially crew and felling supervisors, strike team leaders (dozer, engine and crew), and field observers.

5. Plans for all operational periods should contain clear and specific written instructions regarding line construction and other suppression activities as necessary around cultural values.

Considerations in Prescribed Fires

1. Prescribed burn plans should be designed with a section for cultural clearances, including a signature line for the area archeologist as appropriate.

2. Obtain an on-site inspection and clearance of the burn area.

3. Become familiar with temperatures and durations of the flaming front; know the type and loading of the fuels complex and how they may affect specific cultural materials.

4. Consider the use of chemical foam applications on sites; consult the area archeologist and manufacturer for details and effects.

5. Consider physical removal or reduction of fuels in and around sites.

6. Thoroughly brief all burn personnel on identification and proper protection of cultural features.

7. Consider using archeologist(s) to physically remove

and document artifactual materials from a burn site; this is recommended as a last resort.

8. Include a section on cultural resource protection and clearances in natural prescribed fire plans.

Documentation in Cultural Resource Records

1. Consider incorporating information on fuel build-up and fire vulnerability in cultural resource site record forms or data bases.

2. Following a fire, update site records to document the occurrence of the fire and observed effects on cultural materials.

Research Needs

Far more research is needed on the effects of fire on cultural resources and on effective techniques to avoid or mitigate damage due to fire.

CONCLUSIONS

This paper has attempted to summarize for the manager the nature of cultural resources in the Southwest and how fires might affect them. The effects of fire management activities, including suppression and related activities, have also been addressed, and some ways in which the manager can minimize damage have been presented. The authors hope this information represents just the beginning of a progressively more heightened awareness and commitment to protect these priceless cultural resources.

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Hydrologic and Water Quality Effects of Fire[1]

Malchus B. Baker, Jr.[2]

Abstract.—Prescribed burns usually have minimal hydrologic impact on watersheds because the surface vegetation, litter, and forest floor is only partially burned. Wildfire can, however, have a pronounced effect on basic hydrologic processes, leading to increased sensitivity of the site to eroding forces and to reduced land stability. Fire often causes increased overland flow and greater peak and total discharge, factors responsible for transporting sediment from the site. Fire also causes rapid mineralization and mobilization of nutrients. Because of the natural variability found in forest and range environments in the Southwest, the fire influence continuum, which land managers face in this area, is quite broad.

INTRODUCTION

Water is a valuable resource derived from forests and rangelands, and is also the principal carrier of nutrients through the soil-plant-water-atmospheric continuum. Of all natural resources, water is probably the most sensitive to the disturbance of vegetation and soils on the land surface. Water responses to disturbances include changes in (1) timing and quantity of flow, (2) physical parameters, such as temperature, sediment content, dissolved oxygen, and (3) biological and chemical constituents.

Water yield and stormflow may be increased by burning forest and rangelands. The amount of increase depends on the intensity and severity of burning and the proportion of the watershed burned. Where vegetation is destroyed, interception and evapotranspiration are reduced. Where the organic layers of the forest floor are consumed and mineral soil exposed, infiltration and water storage capacities are reduced. The duration of fire effects ranges from very short periods to many decades, depending on the intensity of the fire itself and rate of vegetation recovery, which is influenced by both natural conditions and remedial measures applied by man.

Occasional fire has always been a natural occurrence in many ecosystems; many existing forest types owe their origin and perpetuation to fire (Stokes and Dieterich 1980). Therefore, what we call normal hydrologic behavior of many forested watersheds already incorporates fire effects.

Prescribed burns are used to attain various timber management objectives and to reduce the risk of insects, disease, and wildfire. These burns are generally made during periods of less intense burning conditions so the damage to the forest floor and understory vegetation is less severe than during intense wildfire. Prescribed burning conditions needed to achieve desired management objectives can, however, be difficult to obtain for the various vegetation and weather conditions found in the Southwest without also causing some adverse effects to the watershed. Burning prescriptions are further complicated by the wide variation in physiographic conditions encountered.

In this review, studies of both prescribed burning and wildfire have been utilized in the assessment of fire effects on the hydrology and water quality from forest and rangeland vegetation types in the Southwest. Although emphasis is intended to be on effects of prescribed burning, often such information was not available. Fire influence in this paper is viewed as a continuum, with effects of prescribed burning at one extreme and wildfire at the other. In a given burning situation, many, if not all, burning conditions will be present to some extent and the final "fire effect" will be an integration of all of them. Therefore, information from situations other than "prescribed burning" is used to estimate possible responses that may be anticipated from burning.

DESCRIPTION OF AREA

As used here, "Southwest" refers to the area between 27° and 37° N. latitude and 103° and 118° W. longitude (Brown 1982). Although this area centers on Arizona and New Mexico, it also includes parts of California, Colorado, Nevada, Texas, Utah, and Mexico (fig. 1). This area supports a range of vegetation that includes major arid and subarid biotic communities.

Vegetation types found in the Southwest, going from lower to higher elevations, are: desert, semi-desert grasslands, chaparral, pinyon-juniper, ponderosa pine, and mixed conifer-aspen (fig. 2). Generally, the higher elevation pine and mixed conifer-aspen types are considered high water and low sediment producing areas (Brown et al. 1974, De-

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[2]*Research Hydrologist, Rocky Mountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287-1304.*

Bano 1977, Hibbert 1979, Hibbert et al. 1974). At mid-elevations, pinyon-juniper and chaparral cover types produce intermediate amounts of water and sediment. The lower lying semidesert and desert areas produce little water and large amounts of sediment.

Precipitation regimes play a major role in the hydrologic response of different vegetation types to burning in the Southwest. Average annual precipitation ranges from about 30 mm to over 800 mm (fig. 2). About 55% of the annual precipitation in the central mountains of Arizona falls as rain or snow between November and April. Snow, a significant factor in the higher montane conifer communities, seldom occurs below the 900-m level in the arid desertscrub vegetation types. Winter storms may release large amounts of water, but their intensities are usually relatively low. Approximately 35% of the annual precipitation occurs during July, August, and September as local convective storms that can be intense and erratic, and that strongly influence erosion and sedimentation. Water yield is essentially nonexistent below an average annual precipitation of 460 mm except for occasional, local, "flashflood" occurrences derived from locally, intense thunderstorms (Hibbert 1979).

Figure 1.—Biogeographic provinces of the Southwest.

ON-SITE EFFECTS

Hydrologic processes that may be affected by fire include precipitation, interception, infiltration and overland flow, soil water storage, snow accumulation and melt, and surface erosion.

Precipitation Interception

Interception—the process of vegetation interrupting the fall of precipitation onto the soil surface—is important because it reduces raindrop impact at the soil surface and, consequently, detachment of soil particles. Soil erosion proceeds in three steps: detachment, transportation, and deposition (sedimentation). The kinetic energy of a falling raindrop on bare soil is sufficient to kick soil particles 1 meter into the air (Hewlett 1982). Osborn (1954b) showed that 50 mm of rain has the potential to detach and set in motion 179 metric

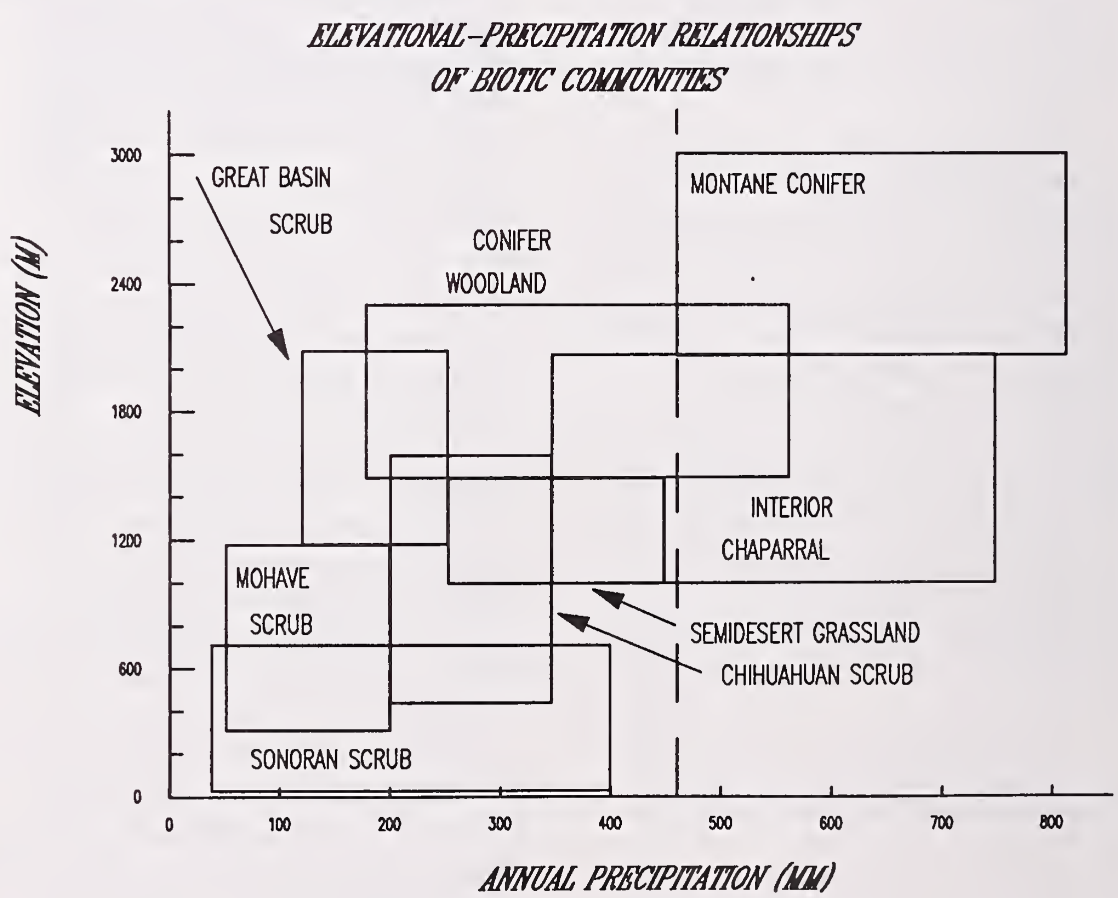

Figure 2.—Elevational-precipitation relationships of biotic communities.

tons of soil material per hectare. The effectiveness of vegetative cover in preventing soil detachment is directly proportional to the amount of cover (Osborn 1954a).

Interception losses during individual rainstorms are a function of interception storage, water stored by plants and the forest floor, and of meteorological conditions, such as precipitation amount, intensity, and duration, and wind speed, during the storm. The percentage of total precipitation lost to interception increases directly with density of vegetation and foliage cover and indirectly with amount of precipitation received and storm duration. Interception losses on undisturbed forest lands during large, flood-producing storms are relatively small, ranging from about 2% to 10%.

The amount of water required to wet vegetation ranges from 0.3 to 2.3 mm for coniferous and hardwood forests (Helvey 1971); for shrubs and grasses it averages 1.3 mm (Zinke 1967). Forest floor interception storage ranges from 2% to 27% of the gross precipitation (Helvey 1971). Zinke (1967) reported that storage values for the forest floor average about 4.1 mm, but other studies indicate the average may be closer to 3.0 mm (Clary and Ffolliott 1969, Garcia and Pase 1967, Kittredge 1955).

Successful prescribed burns in forests consume only part of the forest floor fuels. Prescribed burns do not normally consume canopy material except for some smaller trees in dense stands and possibly occasional scorching of larger trees. In contrast, wildfires often spread through the forest canopy producing an intense fire that not only consumes much of the canopy but also the surface fuels, litter, and forest floor. Thus, prescribed burns have little effect on canopy interception and mainly express their influence by reducing the amount of interception by the lower canopy vegetation and the amount of water storage capacity of the forest floor. Also, because prescribed burns generally produce lower temperatures, they don't have the same effect on the soil properties as wildfires.

In chaparral, differences between prescribed burns and wildfires become smaller because the chaparral canopy carries the fire during both types of burns (DeBano, pers. comm.). There are still some important differences, however, because most prescribed burning is done under marginal or less severe burning conditions when (1) cooler air temperatures and higher humidities prevail, (2) live and dead fuel moistures are high, and (3) wind speeds are low. These conditions often result in a patchy burn, with areas of completely consumed vegetation intermixed with areas partially consumed. Prescribed burns and wildfires in grasslands probably behave much the same way as in chaparral because the canopy (grass cover) also carries the fire in these situations.

Infiltration and Overland Flow

Infiltration may be defined as the amount of water that can move through the soil surface in a given time period. If more water is supplied than can infiltrate, the excess runs off rapidly as overland flow. Some important variables found to affect infiltration under simulated high-intensity rainfall rates (91-127 mm/hr) include: percentage of ground cover, vegetation cover type, soil texture and porosity, and amount of soil organic material, (Dortignac and Love 1961, Meeuwig 1965).

Many of these factors affecting infiltration are adversely affected by fire, resulting in reduced infiltration and increased overland flow (Hendricks and Johnson 1944). Soil organic matter, which is concentrated at or near the soil surface, is directly exposed to heat radiated downward during a fire. Organic matter begins changing chemically when heated to 200°C and is completely destroyed at 450°C (Hosking 1938). Soil organic matter is important for maintaining aggregate stability and soil structure, which affect infiltration and other hydrologic properties of the soil such as water repellency. Fire also indirectly affects microclimate on a site causing greater temperature extremes in both the air and soil (Fowler and Helvey 1978). Such temperature changes can influence soil freezing, soil water depletion, and snowmelt rates which also influence infiltration.

In chaparral vegetation, fire has been shown to create water-repellent layers in the soil (DeBano 1981). Nonwettability increases as intensity of fire increases. These layers create a nonwettable condition that seriously inhibits infiltration and is a major cause of increased overland flow (DeBano 1971, Rice 1974). Although water-repellency problems are most pronounced in chaparral zones, the problem is not confined to this vegetation type.

The effect of fires on runoff is also influenced by burning frequency. Frequent burning can eventually remove most of the protective vegetative cover and, thereby, increase the potential for overland flow.

Prescribed burning.—Prescribed burning probably has its greatest influence, hydrologically, on infiltration and, consequently, on overland flow potential. Because of natural variation in moisture conditions, prescribed burns normally produce a mosaic pattern of consumed organic matter and, consequently, various degrees of influence on water storage capacities and infiltration rates within an area. If properly executed, prescribed burning will not significantly affect, either spatially or temporally, the integrated overland flow and streamflow regime of a watershed.

Overland flow following prescribed burning is directly related to slope steepness and indirectly related to the rate at which disturbed areas are revegetated (Wright et al. 1976).

Vegetation does not usually develop as rapidly or uniformly on steep slopes, as on more moderate slopes.

Light, intense burns can significantly decrease infiltration capacities in the ponderosa pine forest type (Zwolinski 1971). Infiltration data often show a prominent depression after the start of water application caused by soil nonwettability or water repellency. Restoration of the infiltration capacity to near normal often occurs during the winter, however, because of repeated freezing and thawing of the soil. A significant increase in soil bulk density can also be obtained following a heavy burn but not usually after light burning. Removal of organic material probably causes a breakdown in soil structure, resulting in a more compacted surface soil.

Wildfire and slash burning.—Wildfire, as explained, often consumes all or nearly all of the organic matter over extensive areas of a watershed, producing a significant effect on its streamflow regime for a number of years after the fire.

Overland flow on unburned chaparral watersheds rarely exceeds 1% of rainfall and is often nonexistent (Hibbert et al. 1974, Rice 1974). Stormflow increases greatly after wildfire because of increases in surface runoff (overland flow) on severely burned, unprotected, water-repellent soils (Hibbert 1985). In the first year after fire, overland flow has accounted for up to 40% of the rainfall, with the average ranging from 10% to 15% (Rice 1974). Burning in the northern California brush zone normally produces no consistent difference in surface runoff from sparsely covered chamise plots, but runoff may increase by a factor of up to 15 after burning in the denser manzanita, oak, and shrub oak types (Anderson 1949). Postfire recovery is normally rapid, with a decline in severe flooding in 3 years and stormflows in 5 to 10 years (Hibbert 1985).

Water repellency can also develop in sandy textured soils of ponderosa pine ecosystems following wildfire (Campbell et al. 1977). Infiltration rates can be reduced by half (in one example going from 68 to 26 mm per hour), but this situation usually returns to prefire conditions within a matter of years.

Soil Water Storage

In most areas of the United States, the soil mantle is recharged to capacity or near capacity during the springtime. At the start of the growing season, transpiration proceeds rapidly using the readily available water stored in the soil. As the season progresses, water stored in the soil is diminished. Water deficits, that usually exist in the soil by the fall, are subsequently reduced through the winter and early spring. Vegetation removal by fire decreases evapotranspiration, leaving more water in the soil at the end of the growing season than would have occurred if vegetation had been undisturbed.

Prescribed burning.—Fire influences on soil water storage are expressed largely through their effects on evapotranspiration. A properly conducted prescribed burn normally does not greatly affect evapotranspiration and, consequently, little change in the soil water storage is expected.

Wildfire and slash burning.—Wildfire, however, can consume substantial amounts of vegetation, lowering evapotranspiration, and thereby can reduce loss of soil water on a watershed. Minimum soil water content in the fall is often increased compared with prefire conditions. Because of reduced soil water storage capacity, subsequent precipitation is more likely to generate runoff here than on an undisturbed area. A reversed situation has been observed, however, in a severely burned ponderosa pine watershed where increased soil water storage capacity was attributed to greater runoff conditions because of water repellency of the soil and to increased drying of the more exposed soil (Campbell et al. 1977).

Snow Accumulation and Melt

Total snow water equivalent on a watershed at any time throughout the winter is primarily a function of the total snowfall. Additional snow is often deposited directly into small openings in the forest canopy because of increased turbulence (Troendle 1983) or as a savings through the reduction in snow interception by the forest canopy (Baker, in press; Troendle and Meiman 1984, 1986). Important site variables affecting snow accumulation included elevation, aspect, vegetation type, size of trees, the canopy density, and size of openings in the forest canopy. In general, snow accumulation is found to be inversely proportional to vegetation density. Although prescribed fire would have limited effect on canopy interception, openings created by wildfire can influence snow accumulation.

No studies were found documenting the effects of fire on snowmelt rates. However, it is likely that scorching of surface litter and boles of trees would initially increase longwave reradiation to the snowpack, thereby accelerating snowmelt rates even more than those reported in openings created by logging.

Surface Erosion

Erosion in the arid Southwest must be viewed as an unsteady or discontinuous process which transports sediment from a source and through a channel system with intermittent periods of storage (Wolman 1977). This episodic transport process is more characteristic of arid or semiarid climate than of humid regions because the major cause of erosion in the Southwest is the "big"

storm. These big storms move materials from various sources, including the material temporarily stored in the channel system. The disproportionate amount of sediment and debris moved during these major storms makes it difficult to define a "normal rate" of erosion either in the undisturbed or treated condition (DeBano 1977).

Large areas cleared by fires are vulnerable to erosion and can yield substantial amounts of eroded materials if subjected to large, high-intensity summer storms immediately after exposure. Erosion during the winter season is usually less.

Surface erosion, including sheet erosion and rilling, can be defined as the movement of individual soil particles by water or wind; it is a function of available forces, soil surface protection, and the inherent erodibility of the soil. Fire may influence the forces causing erosion by increasing effective precipitation, wind movement, and overland flow.

Fire-induced erosion is greatest in brush and grasslands under a Mediterranean-type climate (frequent droughts and low relative humidity) (Wells 1981). In the 400- to 500-mm annual rainfall region, low soil moisture and poor fertility may delay regrowth and increase erosion. Annual burning often reduces vegetative and litter cover to the point of accelerating erosion, while occasional burning or natural fires usually lead to less long-term erosion unless followed by abusive cultivation, overgrazing, or haphazard timber salvage.

Protection of the soil surface in Arizona chaparral is temporarily reduced by losses of vegetation and surface litter through burning (Pase and Lindenmuth 1971). Soil erodibility is also increased because of the volatilization of soil organic matter and destruction of soil aggregates. The net effect of burning is often an increase in surface erosion (Hibbert et al. 1974, Krammes 1960, Sinclair 1954). However, early establishment of a good grass cover following a fire, and subsequent conservative management, virtually assures soil stability and low sediment yields on moderate slopes (Pase and Granfelt 1977, Wright et al. 1982).

Values shown in table 1 are indications of variability of sediment delivery in response to fire in the Southwest. Documentation of the direct causes of these levels and the great differences among levels has been achieved in only a few studies. Although wildfire has an obvious effect on sedimentation, prescribed burning will generally result in much less sediment production (table 1).

Prescribed burning.—Prescribed burns, by design, do not completely consume extensive areas of organic matter. Therefore, the mosaic pattern produced by restricting the areal extent and degree of water repellency often reduces the amount of soil movement within and from the watershed.

The limited information available on erosion following prescribed burning in chaparral indicates slope, litter, and storm intensity are of major importance (DeBano and Conrad 1976, Pase and Lindenmuth 1971). Erosion increases with slope steepness, the percent of litter removed, and precipitation intensity. High enough intensities can eventually overshadow importance of slope and litter cover.

Studies on ponderosa pine sites in California and Arizona show similar effects of prescribed burning on erosion. In two study areas, a sufficient layer of organic material remained after burning to protect the soil from extensive erosion (Biswell and Schultz 1957, Cooper 1961). In an area in Arizona, erosion occurred only where less than 60% litter cover remained after burning (Pase and Lindenmuth 1971). Although significant increases in the frequency of soil exposure and movement can result after burning, most eroded material only moves a short distance downslope (Cooper 1961).

Wildfire and slash burning.—Intense fires can reduce surface resistance to erosional processes so that critical threshold conditions for soil mobilization and transport are more readily attained. Accelerated erosion appears to result primarily from partial or complete removal of the protective cover (forest floor), leaving the soil surface exposed to the unrestrained erosive forces of raindrop splash and overland flow. Raindrop impact on bare soil directly detaches and moves soil particles over short distances. By dispersing soil fines, splash also tends to seal the surface. This reduces infiltration and promotes overland flow, the process which transports eroded materials over longer distances.

In some areas, notably steep, brush-covered slopes of southern California, fire may increase erosion by loosening surface soil and rocks, causing shallow mass wasting; exposing bare soil to raindrop impact and rill erosion; rendering the top mineral soil layer temporarily nonwettable, thus causing overland flow during the first rain; and sealing the

Table 1.—Effects of fire on sediment delivery.

Author	Vegetation	Location	Treatment	Sediment transport Control	Posttreatment
				kg $ha^{-1}yr^{-1}$	
Glendening et al. (1961)	Chaparral	Arizona	Wildfire	175	204,000
Wright et al. (1976)	Oak-juniper	Texas	Broadcast burn	0.02	28
Biswell and Schultz (1957)	Ponderosa pine	California	Understory burn	0	0
Krammes (1960)	Chaparral	California	Wildfire	5530	55,300
Campbell et al. (1977)	Ponderosa pine	Arizona	Wildfire	0-3	1-1254

surface pores of the soil by splash erosion.

Dry ravel from steep, unburned chaparral slopes can vary from 224 to 4300 kg/ha (Anderson et al. 1959). Dry ravel is accelerated during and immediately after fire, before the rainy season starts. Because dry ravel occurs in the absence of streamflow, debris routinely accumulates in deposits at the base of steep slopes. These deposits, along with untransported remnants of landslide debris, are readily transported downstream when high flood flows occur.

Reported erosion rates that occur in chaparral in the first year after a wildfire range from 9 to 35 times the normal rate, and rates 12 times normal can occur in the second year (Davis 1977, Hibbert 1985, Krammes 1960, Rowe et al. 1954). Largest sources of sediment in chaparral are from scour of residual sediment in the channel, then from rills and gullies, and finally, small quantities from wind, dry ravel, and landslides (Rice 1974). Most of sediment in the channel originates upslope, with landslides and dry ravel as primary contributors.

After the first postfire year, sediment yields usually reflect variation in rainfall and increasing stabilization of slopes and channel bottoms (Pase and Ingebo 1965). Sediment yield from grass-converted watersheds decreases faster than from watersheds that are allowed to recover naturally to chaparral. Slopes on the grass-converted watershed are fairly well stabilized in a matter of years (5 to 10 years) after a fire, and heavy grass cover along the more moist stream channels is often able to anchor much of the deposited sediments. Sediment yields generally drop to near preburn levels within 3 to 5 years (Pase and Ingebo 1965, Pase and Lindenmuth 1971).

Mass soil slippage is usually accelerated on chaparral areas after wildfires in southern California, where slopes are at or near the angle of repose and where rainfall is often heavy. While this type of erosion is not as common in Arizona chaparral, it cannot be excluded entirely as a potential hazard on steep, disturbed slopes (Hibbert et al. 1975).

Substantial erosion (up to 12 tons/ha) can occur after wildfires on conifer watersheds (Campbell et al. 1977, Rich 1962). These sediment yields usually drop to preburn levels in a matter of years.

DOWNSTREAM EFFECTS

The responses of streamflow to on-site effects can result in changes in total annual discharge, peak discharge, stormflows, baseflow, and timing of flow. Water quality and aquatic habitat may also be affected.

Flow Effects

Response to prescribed burning.—There is little information concerning downstream responses below prescribed burned areas. However, because responses below wildfire areas often last only a few years and sometimes only the first runoff season, we can assume that the detection of downstream effects caused by prescribed burning is going to be difficult, if not impossible, to detect. The study of downstream effects on riparian areas, below both wildfire and prescribed burned areas, is definitely needed.

Response to wildfire.—There is little doubt that wildfire can have an influence on downstream environments. As previously discussed, wildfire can have a major influence on vegetation and ground cover resulting in reduced infiltration and increased overland flow. Stormflow increases of threefold to fivefold during the first rainy season are typical in chaparral watershed following a wildfire (Davis 1977, Hibbert 1984, Rowe et al. 1954, Sinclair and Hamilton 1955).

Summer storms in typical, Arizona chaparral account for one-fourth of the annual precipitation, but contribute little to runoff (Hibbert 1984). Even stormflow from intense rain events is generally negligible. However, summer storms after a wildfire can produce a disproportionate amount of stormflow that drops off markedly in a short period of time, often after the first summer season.

In addition to increased stormflows, there is also evidence that wildfires increase baseflow or dry-season flow in chaparral (Colman 1953, Crouse 1961). In Arizona, intermittent streamflow prior to burning can become continuous (Pase and Ingebo 1965). Streamflow will eventually return to an intermittent condition, if the watershed is allowed to revegetate naturally to chaparral. However, if the watershed is converted to grass, perennial streamflow can be sustained.

Along with increases in stormflow and baseflow, there are also responses in peak discharges. Reported increases during the first postfire year in California and Arizona chaparral vary from 2 to 45 times normal, depending on storm size and antecedent moisture conditions (Glendening et al. 1961, Rowe et al. 1954, Sinclair and Hamilton 1955). The time required for peak discharge to return to normal depends on storm size, individual watershed characteristics, and the time needed for the vegetation to reestablish itself. Similar fire effects on peak discharges have been reported in ponderosa pine ecosystems in Arizona (Campbell et al. 1977, Rich 1962).

The hydrologic response to wildfire is documented, but how far downstream does the influence extend? Most of the flow in the Southwest comes during the spring. Therefore, any increases in streamflow because of fire are added to streams that are already flowing. It may be difficult to detect or measure these increases or subsequent losses in flow at any significant distance below a burned area.

Water Quality

Section 208 of the 1972 amendments to the Federal Water Pollution Control Act (Public Law 92-500) specifically mandates identification and control, to the extent possible, of nonpoint-source pollutions resulting from silvicultural activities (USDA Forest Service 1979). The National Forest Management Act (Public Law 94-588), also specifies that land management plans ensure protection of soil and watershed resources.

Sediment

Increased sediment is an important water quality response associated with fire. This subject has been addressed under the section "Surface Erosion."

Fire Influence on Nutrient Losses

Plant communities accumulate and cycle substantial quantities of nutrients in their role as the biological continuum linking soil, water, and atmosphere. Nutrients are cycled in an orderly and predictable manner unless some natural- or human-caused disturbance alters their form or distribution. During a fire, nutrients incorporated in vegetation, litter, and soil can potentially be volatilized during combustion, mineralized during oxidation, or lost by ash convection (Grier 1975). After a fire, nutrients in the ash can be redistributed by wind or leached by water. Part of the plant- and litter-incorporated nutrients (N, P, K, Ca, Mg, Cu, Fe, Mn, and Zn) are volatilized during a fire and removed from the system. Metallic nutrient elements such as Ca, Mg, and K are converted to oxides and deposited as ash on the soil surface. These oxides are relatively insoluble until they react with CO_2 and H_2O of the atmosphere and are converted to bicarbonate salts. These salts are substantially more soluble and vulnerable to loss by surface runoff and leaching. Reductions in plant and litter cover also increase erosion susceptibility of nutrients. Severing the soil-plant cycling mechanisms reduces nutrient uptake opportunities and further increases the potential for nutrient loss by leaching.

Prescribed burning.—Nutrients in small live twigs or dead plant material of chaparral species are either released in a highly soluble form and deposited on the soil surface or are lost by volatilization during fire (DeBano and Conrad 1978). These highly soluble plant nutrients on the soil surface may be used for plant growth or are easily lost by erosion. Increased solubility of nutrients on burned chaparral areas can lead to a lack of a fertilizer response on burned areas (DeBano and Conrad 1976). Above-normal movement of nutrients to streams by surface erosion and leaching has the potential for impairing quality of surface water for municipal purposes, causing eutrophication of aquatic habitats, and lowering of site productivity.

Wildfire and slash burning.—In a study of the effects of fuelwood harvesting and slash burning on nutrient relationships in a pinyon-juniper stand, fuelwood in juniper trees (all material larger than 7.6 cm in diameter) made up about 38% of the tree biomass and contained 25% of the N, 13% of the P, 10% of the K, 25% of the Ca, 12% of the Mg, and 23% of the S in the aboveground tree components (DeBano et al. 1987). Although the leaves made up only 22% of the aboveground biomass, they contained higher percentages of the N, P, K, Mg, and S than the other plant parts. This nutrient distribution is important not only from the standpoint of nutrient removed as fuelwood but also because the nutrients in the slash are exposed to heating when the slash is burned. Slash burning acts as a rapid mineralizing agent for the nutrients contained in the slash, releasing significant quantities of total N, P, K, Ca, and Mg; depositing them on the soil surface.

Bicarbonate Response in Soil Solution and Streamflow

Bicarbonate ions in soil solution and in streamflow are increased as a consequence of burning. The bicarbonate ion is the principal anion in soil solution, is an end product of root respiration in an undisturbed forest, and is a product of oxide conversion following fire. Concomitant fluctuations of bicarbonate and cation concentrations indicate that bicarbonate is the main carrier of cations in the soil solution (Davis 1987).

Nitrogen Response in Soil Solution and Streamflow

Nitrate-N (NO_3), ammonium-N (NH_4-N), and organic-N are the nitrogen forms most commonly studied as indicators of effects of disturbance or land management activity on water quality. Nitrate-N is one of the most mobile ions in soil-water systems and is one of two forms of N used by plants. Maximum recommended levels of NO_3-N in streamflow is 10 mg/L (EPA 1973).

Prescribed burning.—Prescribed burning in California chaparral results in losses of N by volatilization and erosion (DeBano and Conrad 1978). Although this loss represents only 11% (161 kg/ha) of the N in the plants, litter, and upper 10 cm of soil, frequently burned sites would soon be devoid of N if this amount were lost during each fire without a mechanism for replenishing it. We know that precipitation contains some N, although this input amount is probably minimal. A more important input mechanism seems to be nitrogen-fixing organisms. Some shrubs develop root nodules capable of fixing up to 60 kg/ha of N annually under optimum conditions and other postfire leguminous herbs undoubtedly also

fix N (DeBano and Conrad 1978). Nitrogen fixation by nonsymbiotic organisms may also replenish N after a fire, although this source has received little study.

Wildfire and slash burning.—Nitrogen concentrations are normally very low (less than 1 mg/L) in Southwest streams draining undisturbed areas (Davis 1984). Maximum levels of NO_3-N concentration in response to wildfire, alone, range from 0.01 mg/L (Johnson and Needham 1966) to 2.0 mg/L (Longstreth and Patten 1975) (table 2).

Nitrate-N increases in streams appear to be a result of acceleration of the nitrification process in the soil in response to more favorable pH and increased content of electrolytes (mainly Ca) (USDA Forest Service 1979). Nitrate-N easily moves with water through the soil profile to streams. The large increase of 56 mg/L observed in Arizona (table 2) is believed to be in response to NO_3-N release from the massive chaparral root system killed by the herbicide (Hibbert et al. 1974).

Ammonium-N concentration in streamflow can increase an order of magnitude (0.03-0.1 mg/L) after wildfire and concentrations of organic-N about double (Hoffman and Ferreira 1976). Displacement of organic detritus from the stream area because of increased flow and increased stream source area are probably the primary reason for the increased levels of organic-N (Tiedemann et al. 1978).

Phosphorus Response in Soil Solution and Streamflow

Phosphorus in soil solution, streamflow, and lakes is present mainly in two forms—the inorganic ortho-phosphate and organic phosphate as measured by the difference between total phosphate-P and ortho-phosphate-P. In most studies, total phosphate-P is reported as total P (T-P). Even though phosphate-P is an anion, it is not as readily leached as NO_3-N because it complexes readily with organic compounds in the soil (Black 1968).

Phosphorus compounds in pinyon-juniper woodlands are distributed in a mosaic pattern reflecting differences in litter and duff distribution between individual tree species and between tree canopies and interspaces (DeBano and Klopatek 1988). Juniper litter and duff contains higher concentrations of total P and bicarbonate-extractable P than pinyon litter and duff.

Prescribed burning.—Only about 5% (32.1 kg/ha) of the P in California chaparral is contained in the plants and litter (DeBano and Conrad 1978). Almost all P in the plants is returned to the soil surface as ash during prescribed burning because it is concentrated in the smaller plant stems that are easily consumed by the fire. Some P is usually lost by erosion after a burn.

Wildfire and slash burning.—Burning has significant direct and indirect effects on the P cycling process. DeBano and Klopatek (1988) suggest that not only are large amounts of P lost from pinyon-juniper litter during slash burning, but smaller, significant increases in available soil P also occur. About 50% of the total P contained in the litter is lost during combustion. The remaining P in the ash is also susceptible to wind and water erosion, as well as fixation by carbonates.

Phosphorus concentrations have been found to increase in overland flow from slash burning and wildfire areas; but these increases are not usually sufficient to alter quality of stream or lake water (Gifford et al. 1976, Longstreth and Patten 1975).

Cation Response in Soil Solution and Streamflow

Fire substantially alters the form and distribution of cations, making them vulnerable to removal by runoff and leaching. Responses of cations to burning are difficult to interpret because of different amounts in plant biomass and litter, different fire intensities, different exchange capacities of humus and soil, and different moisture fluxes and timing. Studies of soil solutions and surface runoff following fire indicate increases in levels of cations such as Ca, Mg, K, Na, and Mn (Gifford et al. 1976, Sims et al. 1981).

Prescribed burning.—Potassium seems to have a special role in nutrient recycling during fire because a large proportion of this element is contained in the plants and litter. About 73%, or 287 kg/ha, of the K in chaparral is found in the plants and litter (DeBano and Conrad 1978). During prescribed burning, about 15% (44 kg/ha) of the K in the plants is deposited as ash on the soil surface and another 15% is lost—possibly by volatilization. An additional 10% is also lost by erosion and runoff after the fire.

Although 45 kg/ha of Ca, 14.4 kg/ha of Mg, and 5.3 kg/ha of Na are

Table 2.—Effects of fire and selected treatments on maximum NO3-N concentration in streamflow.

Author	Vegetation	Location	Treatment	Maximum NO_3-N Control	Maximum NO_3-N Posttreatment
Hibbert et al. (1974)	Chaparral	Arizona	Herbicide, fire	0.20	56.00
Johnson and Needham (1966)	White-fir Ponderosa pine	California	Wildfire	0.01	0.01
Hoffman and Ferreira (1976)	Mixed conifer shrub	Sierra Nevada	Wildfire	0.06	0.30
Longstreth and Patten (1975)	Chaparral	Arizona	Wildfire, maintained in grass	0.10	2.00

translocated to the soil surface during prescribed burning in chaparral, their importance in nutrient cycling in this ecosystem is not fully known (DeBano and Conrad 1978). About 67 kg/ha of Ca, 32 kg/ha Mg, and 4.6 kg/ha of Na are also lost by erosion and runoff. This condition suggests that not only are soluble cations deposited in ash lost following burning but that some of these elements in the burned and unburned litter can also be eroded away.

Wildfire and slash burning.—Watershed studies provide an integrated view of the effects of fire on cation concentrations and losses. Johnson and Needham (1966) conducted the first watershed study of effects of fire on chemical water quality. They found no pronounced effect of wildfire on ionic composition and concluded that increased runoff resulting from cover reductions masked concentration effects. Longstreth and Patten (1975), however, found that Ca and K concentrations can increase after burning in chaparral.

Sediment Losses of Nutrients

Sediment losses of N, P, and cations in California chaparral can substantially exceed those lost in solution after a wildfire (DeBano and Conrad 1976). Nitrogen and P losses of 15.1 and 3.4 kg/ha, respectively, are reported in sediments as compared with only trace amounts found in solution. Loss of Ca, Mg, Na, and K in solution is about one-fourth of the loss on sediment. Overall, however, sediment and solution losses of nutrients often comprise only a minor proportion (0.7-8%) of the total prefire nutrient capital of plants, litter, and upper 10 cm of soil for N, P, K, Mg, Ca, and Na.

Aquatic Habitat Responses

Although there is an accumulating data base of effects of fire on nutrients and stream water chemistry, attendant responses at the stream level have not been well studied. Hoffman and Ferreira (1976) examined periphytic algae above and below burned sites in California and found essentially no difference in the similarity index, indicating that water quality changes did not exert any measurable effect on algae growth.

MANAGEMENT IMPLICATIONS

Fire can be used economically in managing and manipulating vegetation, often in an ecologically sound manner. Mechanical means of preparing seedbeds are expensive, and spraying with herbicides is not only expensive but also controversial. Initially, however, fire may need to be used in combination with other improvement methods.

Costs of fire suppression are increasing every year. Risk of wildfire can be reduced by the use of prescribed burning. A study of 26,000 ha of prescribed burn in Arizona showed that 82% fewer fires occurred in this area in the 3 years following control burning (Hedden 1957). Prescribed burning in some forest situations will result in some pruning of the lower tree branches and some reduction of stems and fuelwood. These factors often contribute to increases in environmental esthetics, better accessibility for recreational uses, increases in nutrient availability for plant growth, improved wildlife habitat, and a reduction in insect and disease damage.

Most land managers in the Southwest have responsibility for at least two or three vegetation types, and the influence of burning in these different ecosystems are usually different. Considerable information is available concerning hydrologic response to burning, particularly following wildfire. Generally, hydrologic responses after burning change proportionally with precipitation and are minor where annual precipitation is less than 460 mm (Hibbert 1979). This precipitation criterion applies to all desertscrub types, the semidesert grasslands, and much of the conifer woodland type (fig. 2). Therefore, only the montane conifer, much of the interior chaparral, and some of the conifer woodland type with annual precipitation regimes above 460 mm have the potential of exhibiting much of a hydrologic response to burning.

Because of the natural variability found in forest and range environments in the Southwest and in burning situations, fire influences are viewed as a continuum, with effects of prescribed burning at one extreme and wildfire at the other. Each burning situation, whether a prescribed burn or wildfire, will result in a mosaic pattern of site conditions. Land managers, consequently, must often deal with a wide spectrum of fire influence on a given burned area. However, with increased knowledge and its implementation, perhaps someday more time and money can be expended on the prescribed burning end of the scale.

Light, prescribed burns usually have minimal hydrologic impact on watersheds as the result of partial burning of the surface vegetation and litter and of the forest floor. Wildfire, however, can kill trees and other vegetation and can consume the forest floor over large areas of a watershed. As a result, wildfire can exert a pronounced effect on basic hydrologic processes, leading to increased sensitivity of the site to eroding forces and to reduced land stability. Response to fire is often exhibited as an increase in overland flow and an increase in peak and total stream discharge. These factors, consequently, provide the transport forces for removing sediment and nutrients from the site.

Erosion, in response to burning, is a function of several factors including degree of protective cover loss; steepness of slopes; degree of soil nonwettability; climatic characteris-

tics; and rapidity of vegetation recovery. Resulting sedimentation and increased turbidity appear to be the most serious threats to water resources following fire.

Most of the streamflow in the Southwest comes during the spring runoff period. Therefore, any increases due to burning will normally be added to flowing streams. Consequently, it may be difficult to detect or measure any effect of fire even below severely burned areas. Most evidence suggests that even after wildfires, responses in streamflow and sediment yields drop to near preburn levels within 3 to 5 years.

The influence of burning on nutrients is not as clear. Fire causes rapid mineralization and mobilization of nutrient elements that are manifested as increased levels of nutrients in overland flow and in soil solution. Burning can also result in the loss of various amounts of some elements by volitalization and convection. Studies indicate that additional nutrients in streamflow after burning do not significantly impair the quality of surface waters for municipal purposes, but more information is needed on this subject and on their effects on the riparian community.

Often nutrient losses following fire are not large compared with the total amount of nutrients on the site. However, what effect the periodic losses of nutrients have on "watershed condition" or what "cumulative effects" these losses have on site productivity are not well established.

Prescribed burning can be used as a management tool in the various vegetation types found in the Southwest. However, land managers must always be aware of the delicate balance that exists within the soil-plant-water-atmosphere system and how easily it can be, either positively or negatively, influenced by fire. Two major gaps in our knowledge are the cumulative effects of burning on watershed condition, and the effects of burning on riparian habitat. These two areas must be studied before the full potential of prescribed burning can be realized.

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Fire Effects On Grasses In Semiarid Deserts[1]

Jerry R. Cox, F.A. Ibarra-F, M.H. Martin-R[2]

Abstract.—Land managers have recommended fall and winter burning of semidesert grasslands for 100 years, but historical evidence indicates that the greatest potential for natural fires would have occurred in summer. The purpose of this paper was to (1) determine how seasonal burning influences buffelgrass *(Cenchrus ciliaris)*, Lehmann lovegrass *(Eragrostis lehmanniana)*, and big sacaton *(Sporobolus wrightii)* herbage production and (2) assess the importance of fire in the management of semiarid deserts.

Semidesert grasslands and shrublands in North America are bordered on the east by Rio Grande River, on the west by the Colorado River, on the north by the Colorado Plateau, and on the south by the Sierra Madre in Mexico. In this area, lightning strikes from summer thunderstorms have started fires since Pleistocene (Pyne 1982), but when European man entered the area fire was applied in spring, fall and winter (Bahre 1986; Hastings and Turner 1972).

Seasonal burning kills tree and shrub seedlings and has little or no effect on perennial grasses in the American tall-grass prairies (Albertson and Weaver 1945; Weaver and Rowland 1952). Hence the assumption by Sauer (1944), Humphrey (1958) and Wright and Bailey (1982) that fire has no effect on semidesert grasses. In the past 10 years we have burned native and introduced semidesert grasslands to determine seasonal fire effects on perennial grasses. The purpose of this paper is to summarize our findings and place a practical value on fire in semidesert grasslands and shrublands.

Study Sites

Three sites were selected along an elevational gradient between southeastern Arizona and northern Sonora, Mexico. Sites were located (1) 120 km north of Hermosillo, Mexico (Carbo); (2) 40 km south of Tucson, Arizona (Santa Rita Experimental Range-SRER); and (3) 80 km southeast of Tucson (Empire Ranch). Predominant perennial forage grasses were buffelgrass *(Cenchrus ciliaris)* at Carbo, Lehmann lovegrass *(Eragrostis lehmanniana)* at SRER, and big sacaton *(Sporobolus wrightii)* at Empire Ranch.

Precipitation, temperature, and soil classification are presented in table 1. Elevation is greatest at Empire Ranch, intermediate at SRER and least at Carbo. Precipitation distribution is approximately 60% in summer and 40% in winter. Freezing temperatures are uncommon at Carbo but do occur in January and February. Freezing temperatures occur between December and March at SRER, and between November and April at Empire Ranch.

Methods And Materials

Buffelgrass Herbage Sampling

A 15-ha study area was fenced within a 750-ha buffelgrass pasture at Carbo, Mexico. Three, 50 by 100 m

Table 1.—Site characteristic and soil classification at three burning sites.

Sites	Elev.	Precipitation Summer	Precipitation Winter	Daily temp. Max.	Daily temp. Min.	Soil series	Family classification
	m	——mm——		——°C——			
Carbo	600	175	100	41	4	Anthony	Fine - loamy, mixed, thermic, Typic Torrifluvent
SRER	1,050	200	125	33	3	Comoro	Coarse - loamy, mixed, thermic, Typic Torrifluvent
Empire Ranch	1,370	250	150	33	-2	Pima	Fine-loamy, mixed, thermic, Typic Haplustoll

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[2]Cox is range scientist, USDA, Agricultural Research Service, Aridland Watershed Management Research Unit, 2000 E. Allen Road, Tucson, Ariz. 85719; Ibarra-F and Martin-R are technical director and head of the Department of Range Management, respectively, CIPES, Periferico OTS S/N, Conjunto UGRS, Postal 1754, Hermosillo, Sonora, Mexico.

plots were burned with a head fire in either summer (July) or fall (October) 1985, 1986, and 1987; plots were burned once. Ten, 1 by 1 m sampling areas were randomly selected for sampling 1, 2, and 3 years after the fall burns. Buffelgrass was harvested at the soil surface and separated into green and dead herbage. Herbage was dried at 40°C for 72 h, and weighted. Herbage components from the ten sampling areas in a plot were averaged, and the mean considered a replication.

Lehmann Lovegrass Herbage Sampling

Forty-five, 15 by 15 m plots were marked in a 6-ha exclosure. Plots were burned with a head fire in either winter (February), spring (June), summer (July) or fall (October) 1984, 1985 and 1986. Ten, 0.25 by 0.25 m sampling areas were randomly selected for sampling 1, 2, and 3 years after the fall burns. Herbage was harvested at the soil surface, and three sampling areas from a plot were separated into green and dead herbage. Separated and unseparated herbage samples were dried at 40°C for 72 h and reweighed.

Either green or dead dry weights from the three separated sampling areas in a plot were summed, and the contribution of each to the total dry herbage expressed as a percentage. Component percentages were multiplied by the total herbage dry weight. The derived dry weight component values for a plot were averaged and the mean considered a replication.

Big Sacaton Herbage Sampling

In January 1980, 15 by 15 m plots were marked within a fenced 2-ha big sacaton study site. Plots were burned with a head fire in winter (February), summer (July) and fall (October) 1980, 1981, and 1982. Twenty, 0.3 by 2.9 m sampling areas were randomly selected for sampling 1, 2, and 3years after fall burning. Plants were harvested at the soil surface in four sampling areas and herbage separated into green and dead. Separated samples were weighed in the field and a modified weight-estimate technique used to estimate both green and dead herbage (Pechanec and Pickford 1937) in 16 sampling areas. Separated samples were dried at 40°C for 48 h and weighed. Regression techniques were used to estimate dry weights from estimated field weights (Campbell and Cassady 1949).

Design and Statistical Analysis

Experimental design was a completely randomized block with either three (SRER) or four (Carbo) replications and burning treatments applied in either two (Carbo) or three (SRER) seasons during three consecutive years. At Empire Ranch, treatments were arranged in a split block design with three replications. The three years were randomized as blocks within a replication and the three seasons were randomized within years (Cox 1984, 1985).

To demonstrate long-term fire affects we are presenting herbage data collected in August or September, one, two, and three growing seasons following fall treatments. Precipitation amounts, grasses, statistical design and data collection techniques differ among sites. Therefore, statistical comparisons are site specific. General comparisons, however, will be made among sites.

Fire Temperatures

In Lehmann lovegrass plots burned in spring, summer and fall 1985 and 1986, and in buffelgrass plots burned in summer 1986 fire temperatures were recorded at 5 sec. intervals with a Campbell Scientific 21 X data logger. In each plot, surface temperatures were measured in six crowns and in six open areas between plants. Temperatures were similar at crown and open locations, therefore, temperature for surface locations were combined.

Results

Buffelgrass Herbage

Summer precipitation was 186, 311 and 144 mm (fig. 1) in 1985, 1986 and 1987, respectively. Green leaves appeared in less than 10 days on all

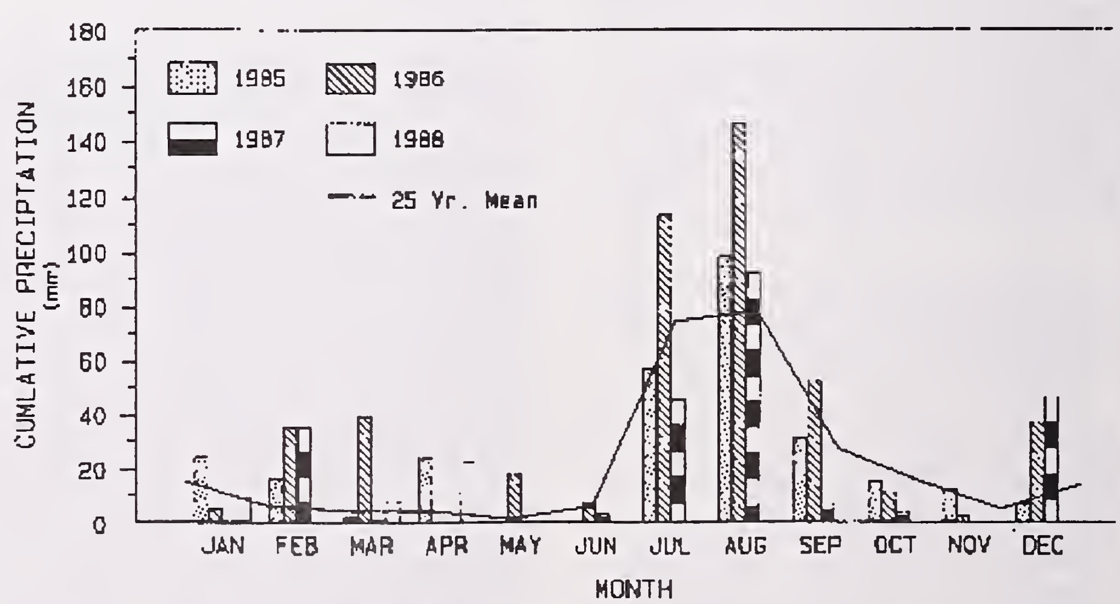

Figure 1.—Monthly precipitation amounts from 1985 to 1988 at a buffelgrass site near Carbo, Sonora, Mexico, and the monthly 25-year means at Carbo.

buffelgrass plants burned in summer; but growth in the following 60 days depended on rainfall amount and distribution. In summer 1986, two or three storms occurred weekly during July and August, weekly amounts exceeded 25 mm, and in mid-September green herbage from summer burned areas (1,900 kg/ha) was 25% greater than unburned areas. While in summers 1985 and 1987, storms were widely spaced, weekly amounts varied from 0 to 40 mm, and in mid-September green herbage from unburned areas (300 kg/ha) was equal to summer burned areas.

Fall, winter and spring precipitation was 113, 86 and 85 mm (fig. 1) in 1985-86, 1986-87 and 1987-88, respectively. Fall burned plants produced green leaves in either December (1986 and 1987) or February (1986), but active growth occurred only in March 1986. Cool season moisture between February 20 and March 10, 1986 totalled 75 mm, and buffelgrass plants in burned and unburned areas produced large quantities of green leaves when nighttime temperatures exceeded 10°C in March. In March 1985 and 1987, growth was limited by dry soils and cold nighttime temperatures.

In the first summer growing season after burning in 1985 (1986), buffelgrass green herbage from unburned plants was 50% greater than that collected from burned plants (table 2). While green herbage collected in summer 1987 (1 year after the 1986 burns and 2 years after the 1985 burns) varied from 200 to 555 kg/ha, it was similar among years and treatments. Green herbage produced in spring and summer becomes dead herbage in fall (table 3). Burning, irrespective of season, removes dead herbage. Dead herbage is not an ideal forage resource because crude protein is less than that needed by lactating cows, but dead herbage removal must be followed by a reduction in stocking rates (Cox and Morton 1986).

Table 2.—Buffelgrass green forage at the peak of the summer growing season (September 15), 1, 2, and 3 years after burning.[1]

Number of growing seasons after treatment	Season of treatment	Treatment year 1985	1986	1987	LSD
		————kg/ha————			
1	Summer burning	1,030	465	800	235
	Fall burning	970	225	350	
	Untreated	2,070	280	1,075	
2	Summer burning	555	1,375	[2]	
	Fall burning	202	1,000	[2]	
	Untreated	270	1,200	[2]	
3	Summer burning	750	[2]	[2]	
	Fall burning	400	[2]	[2]	
	Untreated	1,000	[2]	[2]	

[1]*Data were collected in September, following the fall treatments.*

[2]*Data to be collected in September 1989 and 1990.*

Table 3.—Buffelgrass dead forage at the peak of the summer growing season (September 15), 1, 2, and 3 years after burning.[1]

Number of growing seasons after treatment	Season of treatment	Treatment year 1985	1986	1987	LSD
		————kg/ha————			
1	Summer burning	1,030	416	1,200 [2]	460
	Fall burning	600	710	800	
	Untreated	2,425	2,515	2,400	480
2	Summer burning	2,435	1,200	[3]	
	Fall burning	1,375	895	[3]	
	Untreated	3,125	3,000	[3]	
3	Summer burning	3,000	[3]	[3]	
	Fall burning	3,200	[3]	[3]	
	Untreated	3,180	[3]	[3]	

[1]*Data were collected in September, following the fall treatments.*

[2]*When a bracket includes only treatments, the variances between treated and untreated means were not homogeneous. Therefore, a separate ANOVA was used for each set. Means in each set that differ more than the given Least Significant Difference value are significantly different at $P \leq 0.05$.*

[3]*Data to be collected in September 1989 and 1990.*

Lehmann Lovegrass Herbage

Summer (June-August) precipitation at the Lehmann lovegrass site was 115, 5, 10 and 60% greater than the long-term mean (169 mm) in 1984, 1986, 1987 and 1988, respectively, but 25% less than the long-term mean in 1985 (fig. 2). Cool-season (September - April) precipitation which included 3 to 5 snow storms in December and January was 125 to 200% greater than the long-term mean (147 mm), in the three years.

Lehmann lovegrass leaves appeared within 14 days after burning in winter, spring, summer and fall,

but green leaves on fall burned plants disappeared in October and November when nighttime temperatures varied from 5° to 10°C. Green leaves in winter, spring and summer burned areas, and unburned areas remained until nighttime temperature dropped below 5°C in mid-December. Observations indicate that below freezing temperatures kill Lehmann lovegrass plants burned in fall, but fall fires stimulate seed germination in the following summer (Ruyle et al. 1988). In summer following a fall burn, seedling densities varied from 50 to 300/m^2.

In the first summer growing season after the 1984 and 1986 treatment years, Lehmann lovegrass green herbage on fall burned areas was less than in winter, spring and summer burned, and unburned areas (table 4). By the second and third growing seasons, green forage on burned areas equaled unburned areas.

In Lehmann lovegrass stands, 60% of the green herbage produced in summer can transfer to dead herbage in 5 to 10 days. Therefore, total above-ground herbage (table 4, 5) is a better estimator of annual production. In the first and second summer growing seasons total above-ground herbage on fall burned areas was less than on other burned areas. However, total production on fall burned areas exceeded or equaled that on winter, spring and summer burned areas 3 years after 1984 and 1985 treatments.

Big Sacaton Herbage

Summer (June-August) precipitation was 125, 173, 128, 142, 302 and 165 mm (fig. 3) in the consecutive years between 1980 and 1985; while cool-season (September-April) precipitation was 250, 160, 525, 960 and 279 mm. Summer precipitation exceeded the long-term mean (225mm) only in 1984, but cool-season precipitation exceeded the mean (205mm) in all years except 1981-82.

Big sacaton leaves began to appear within 20 days on plants burned in winter and within 3 days on plants burned in summer (Cox 1988). Burning in either winter or summer appeared to stimulate green leaf production in August. Green leaf production in August (24 weeks after winter and 6 weeks after summer burns), however, was consistently

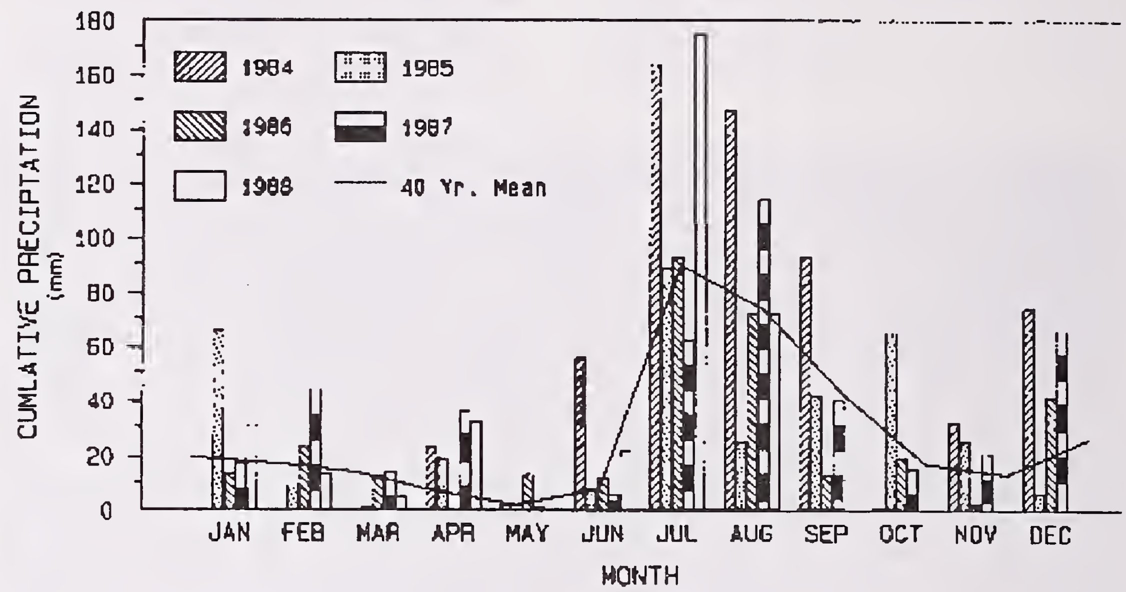

Figure 2.—Monthly precipitation amounts from 1984 to 1988 at a Lehmann lovegrass site in the Santa Rita Experimental Range (SRER), and the monthly 40-year means at Florida Canyon, SRER.

Table 4.—Lehmann lovegrass green forage at the peak of the summer growing season (August 30), 1, 2, and 3 years after burning.[1]

Number of growing seasons after treatment	Season of treatment	Treatment year 1984	1985	1986	LSD
		———kg/ha———			
1	Winter burning	1,515	1,245	1,195	[2]
	Spring burning	1,290	1,145	1,110	
	Summer burning	1,385	940	1,084	240
	Fall burning	885	1,295	955	
	Untreated	1,520	1,720	1,115	
2	Winter burning	1,265	950	1,285	
	Spring burning	1,075	915	1,230	
	Summer burning	1,100	1,025	1,365	260
	Fall burning	1,050	1,185	1,190	
	Untreated	1,305	1,025	950	
3	Winter burning	755	1,260	[3]	
	Spring burning	920	1,005	[3]	
	Summer burning	680	870	[3]	390
	Fall burning	960	1,220	[3]	
	Untreated	780	1,175	[3]	

[1]Data were collected in August, following the fall treatments.

[2]Means in each set (enclosed within a bracket) that differ more than the given Least Significant Difference value are significantly different at $P \leq 0.05$.

[3]Data to be collected in August 1989.

greater on unburned areas, intermediate on winter burned areas, and less on summer burned areas.

In the first and second growing seasons after treatment, big sacaton green herbage accumulated more rapidly on summer than on winter or fall burns (table 6), but differences among seasons and treatments occurred infrequently. By the third summer growing season, green herbage on summer burns was usually greater than on winter and fall burns.

In the second and third summer growing seasons, big sacaton seed stalks accumulated more rapidly on summer burns than on winter and fall burns (Cox 1988). Therefore, dead herbage quantities on summer burns exceeded those on winter and fall burns (table 7).

Table 5.—Lehmann lovegrass dead forage at the peak of the summer growing season (August 30), 1, 2, and 3 years after burning.[1]

Number of growing seasons after treatment	Season of treatment	Treatment year 1984	1985	1986	LSD
		————kg/ha————			
1	Winter burning	1,785	1,185	1,245	[2]
	Spring burning	955	625	965	
	Summer burning	1,365	445	770	410
	Fall burning	10	235	115	
	Untreated	2,280	1,845	2,170	
2	Winter burning	2,215	2,575	1,885	
	Spring burning	1,930	1,905	1,685	
	Summer burning	1,805	2,250	2,480	450
	Fall burning	1,125	670	1,055	
	Untreated	2,330	2,475	2,740	
3	Winter burning	2,655	2,310	[3]	
	Spring burning	2,245	2,325	[3]	
	Summer burning	2,150	2,060	[3]	485
	Fall burning	2,790	1,875	[3]	
	Untreated	2,755	3,215	[3]	

[1]Data were collected in August, following the fall treatments.

[2]Means in each set (enclosed within a bracket) that differ more than the given Least Significant Difference value are significantly different at $P \leq 0.05$.

[3]Data to be collected in August 1989.

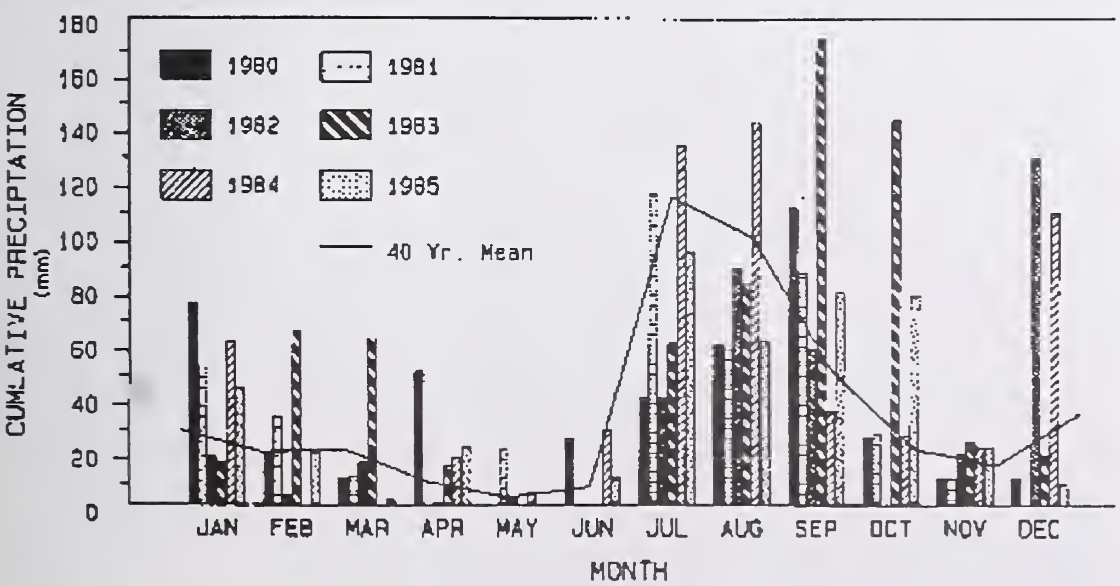

Figure 3.—Monthly precipitation amounts from 1980 to 1985 at a big sacaton site (Empire Ranch) near Sonoita, Arizona, and the monthly 40-year means at Patagonia.

Fire Temperatures

Temperatures in spring, summer and fall (figs. 4 and 5) head-fires were highly variable among years, plots and locations within plots. Fire temperatures at the soil surface, irrespective of season, were highest (300-585°C) when herbage was dry and lowest (100-275°C) when herbage was either green or moist.

Discussion and Conclusions

Before conducting a controlled burn the land manager should consult Wright and Bailey (1982). Instructions in this source are clearly presented and easy to follow. The manager should remember, however, that burning at any season removes buffelgrass, Lehmann lovegrass and big sacaton phytomass and exposes the crown, but fall defoliations leave the crown exposed in winter when temperatures are frequently near or below freezing. Fall defoliations (1) remove small quanties of green herbage available to livestock in fall and winter, and (2) reduce green herbage availability in early spring. The result in Lehmann lovegrass and big sacaton grasslands is the complete loss of the grazing resource for 200 to 245 days after fall defoliation.

Burning at any season removes green buffelgrass, Lehmann lovegrass and big sacaton herbage available to livestock and reduces the amount of green herbage production for at least the next 2 years. We believe these results accurately represent conditions in the Southwest because data were collected in summers when precipitation was either above, below or equal to the long-term mean.

The dead herbage of the three grasses is not an ideal grazing resource because crude protein content annually varies from 3-7%, but dead herbage removal influences pasture stocking rate. Cox and Morton (1986) and Cox et al. (1989) have shown that defoliations before or during winter cause a three-fold decrease in spring-summer stocking rates, while steer gains on burned pastures were only one-third greater than on unburned pastures.

Management Implications

The studies presented in this paper include time and space replications, but the data is limited in value if not interpreted in "real world" application. Therefore, we will attempt to overlook our own preconceived notions and accurately assess the importance of fire in semiarid deserts.

Current fire management authorities formulated their ideas from scientifically conducted studies at grassland sites to the east and north of the semiarid deserts. Climatic conditions at those sites were usually cooler and wetter but as in the semiarid desert, rainfall during active plant growth was below normal in 3 or 4 of 10 years. Authorities, however, failed to realize that drought conditions at cooler and wetter sites were less severe than ideal conditions in semiarid deserts. Hence perennial grasses burned at eastern and northern grassland sites have greater recovery opportunities under drought conditions than burned semiarid desert grasses under ideal conditions. Therefore, land managers in southern Arizona, southwestern New Mexico, northwestern Chihuahua and Sonora can expect a fire to adversely affect perennial grass herbage production for 2 or 3 years.

Scientific studies at eastern and northern grassland sites also show that fire (1) improves forage quality and (2) kills undesirable trees and shrubs. Managers in semiarid desert have reported similar results, but our studies do not support their conclusions. In the past 10 years we have conducted more than 150 control burns in semiarid deserts, and following 87 of these fires we measured forage quality at two week intervals for 1 year. Green herbage quality on 64 burns exceeded that on unburned

Table 6.—Big sacaton green forage at the peak of the summer growing season (August 21), 1, 2, and 3 years after burning (Cox 1988).[1]

Number of growing seasons after treatment	Season of treatment	Treatment year 1980	1981	1982	LSD
		————kg/ha————			
1	Winter burning	735	615	400	[2]
	Summer burning	865	770	700	205
	Fall burning	725	665	260	
	Untreated	1,900	2,600	1,695	535
2	Winter burning	850	745	690	
	Summer burning	925	935	850	240
	Fall burning	805	755	575	
	Untreated	1,365	1,720	1,705	
3	Winter burning	615	595	695	
	Summer burning	1,590	1,450	1,375	525
	Fall burning	745	890	790	
	Untreated	1,650	1,845	2,000	

[1]Data were collected in August, following the fall treatments.

[2]Means in each set (enclosed within a bracket) that differ more than the given Least Significant Difference value are significantly different at $P \leq 0.05$.

Table 7. Big sacaton dead forage at the peak of the summer growing season (August 21), 1, 2, and 3 years after burning (Cox 1988).[1]

Number of growing seasons after treatment	Season of treatment	Treatment year 1980	1981	1982	LSD
		————kg/ha————			
1	Winter burning	0	0	0	[2]
	Summer burning	0	0	0	
	Fall burning	0	0	0	
	Untreated	3,080	1,000	2,305	780
2	Winter burning	1,220	320	145	
	Summer burning	1,935	1,180	1,020	685
	Fall burning	650	445	100	
	Untreated	2,710	1,975	1,635	
3	Winter burning	1,650	1,145	1,430	
	Summer burning	2,035	1,665	1,840	735
	Fall burning	570	690	850	
	Untreated	1,980	1,290	1,970	

[1]Data were collected in August, following the fall treatments.

[2]Means in each set (enclosed within a bracket) that differ more than the given Least Significant Difference value are significantly different at $P \leq 0.05$.

areas for 14 days or less, while on 15 burns quality was enhanced for more than 14 but less than 28 days. On the remaining burns (8) quality was enhanced for more than 28 days but less than 42 days. Using fire to improve forage quality in semiarid deserts is difficult to justify when quality improvements are short lived and herbage production is adversely affected for 2 or 3 years.

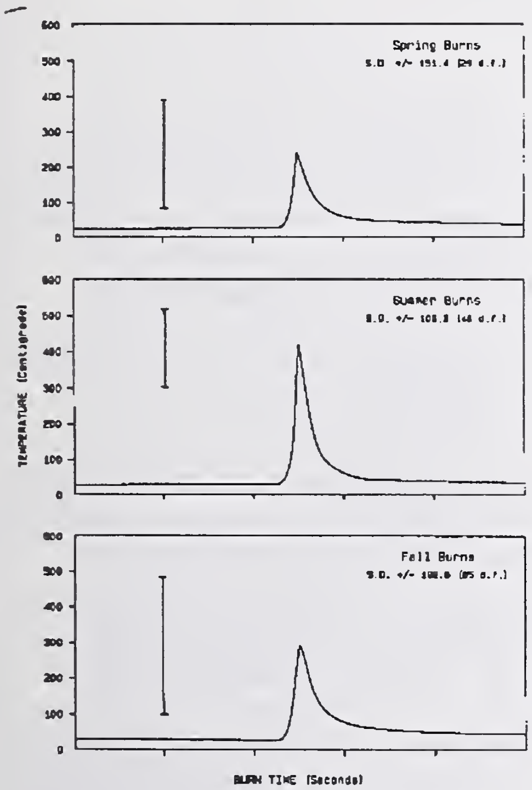

Figure 4.—Headfire temperatures in Lehmann lovegrass stands at Santa Rita Experimental Range, during spring, summer and fall 1985 and 1986.

Thirty-two burned areas were intentionally reburned either 1, 2 or 3 years after treatment, and eight areas were reburned by wildfires. Only on multiple burned areas, have we measured a 50% mesquite mortality. Mesquite burned once may be totally defoliated, but plants resprout within 36 months. Therefore, multiple burning strategies to reduce mature mesquite populations and kill seedlings will be necessary if the semiarid grasslands are to return.

Cattle need herbage to exist and fires reduce herbage production. It is our opinion that the two are incompatible under semiarid conditions. Therefore, the semiarid shrublands will remain indefinitely where grazing continues. Where grazing has been discontinued, fire can be used to reduce shrub populations and stimulate the return of semiarid grasslands. When perennial shrubs dominate a site, livestock removal will not be followed by an increase in perennial grass herbage production. Therefore, brush populations must be reduced before fire can be used as a management tool.

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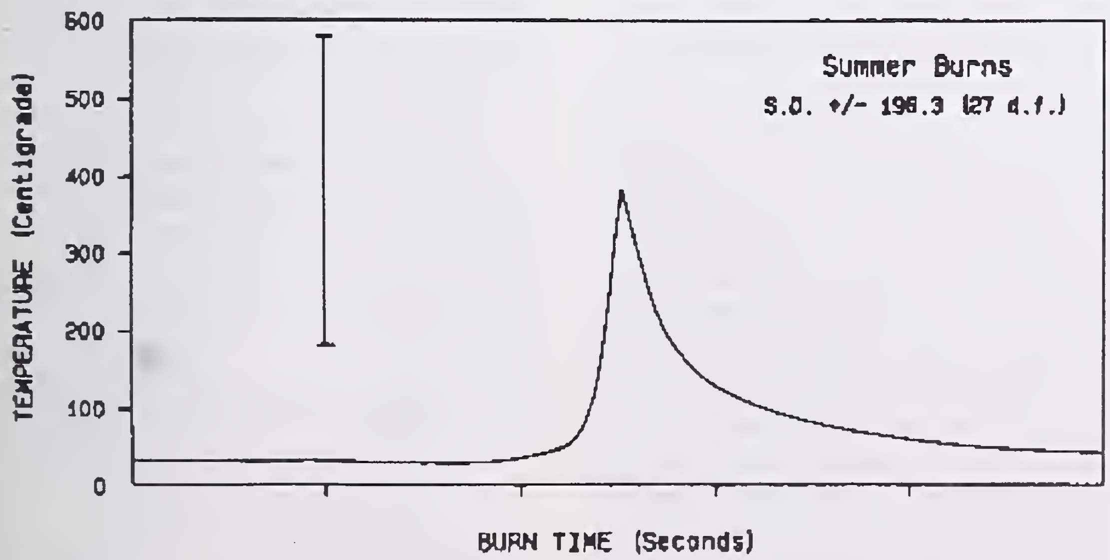

Figure 5.—Headfire temperatures in buffelgrass stands near Carbo, Sonora, Mexico, during summer 1986.

Effects of Fire on Wildlife in Southwestern Lowland Habitats[1]

Carl E. Bock[2] and Jane H. Bock[2]

Abstract.—Prescribed burning can benefit most wildlife in Great Basin shrubsteppe, chaparral, and semidesert grassland, especially if it is used to create fine-scale mosaics of native vegetation, including some unburned stands. Fire is likely to negatively affect vegetation and wildlife in Sonoran desertscrub, Chihuahuan Desert grassland, and riparian woodland.

The purpose of this paper is to review and synthesize information about the responses of wildlife to fire in the grasslands and shrublands of Arizona and New Mexico. We begin by describing the effects of one particular fire at a site we have been studying intermittently since 1974. We intend this as more than personal indulgence. Rather, the wildlife responses to this fire, and to others we have studied at the same site, lead us to certain conclusions that we next will test for lowland southwestern habitats generally.

On July 14, 1987, lightning ignited a wildfire on the Appleton-Whittell Research Ranch, a National Audubon Society sanctuary in the grasslands and oak savannahs of southeastern Arizona. Summer rains were late that year, and in fact they had not yet begun to any significant extent. Atmospheric conditions were hot, dry, and very windy. There was a great deal to burn, since the area had not been grazed by livestock since 1968 (Bock et al. 1984). U. S. Forest Service fire crews, with the assistance of local volunteers, were able to contain the burn by the evening of July 15, largely along firebreaks such as the sanctuary's boundaries with adjacent operating cattle ranches. By this time, the fire had completely blackened the grasses, shrubs, and scattered mesquite on about one-third of the sanctuary's 3200 ha.

Summer rains began four days after the fire, followed quickly by dramatic vegetation and wildlife responses. Because we had worked on the sanctuary prior to this burn, on sites spared as well as those combusted by the fire (Bock et al. 1986), we were in a position to accurately measure its effects upon plant and animal populations. At the end of the 1987 growing season, burned study plots supported about one-third the grass cover, less than half the shrubs, and over twice the herb cover as that found on nearby unburned areas. Insect populations were reduced, and seed production was dramatically higher on the burn. Certain animal species requiring heavy cover, such as the cottonrat *(Sigmodon hispidus)*, Botteri's sparrow *(Aimophila botterii)*, Cassin's Sparrow *(A. cassinii)*, and grasshopper sparrow *(Ammodramus savannarum)*, disappeared entirely from the burned areas. Overall, the wildlife response was positive. Mule deer *(Odocoileus hemionus)* and pronghorn *(Antilocapra americana)* used burned areas more frequently than comparable unburned sites through the fall and winter of 1987-88. Mourning doves *(Zenaida macroura)* concentrated on the burn in that first post-fire year in densities nearly fifteen times higher than those typical of unburned grassland. Birds collectively were more than five times more abundant on the burn than they were either on adjacent control plots, or on our burned plots in years prior to the fire.

By the end of the 1988 growing season, grass cover on burned plots was nearly 75% that on control areas, while herb cover remained more than two times greater on the burn. Shrubs were reduced, due largely to near total fire-kill of burro weed *(Haplopappus tenuisectus)*. Cassin's, Botteri's, and grasshopper sparrows had begun to re-colonize the burn, but in numbers far lower than nested in unburned sites. Horned larks *(Eremophila alpestris)*, lark sparrows *(Chondestes grammacus)*, and mourning doves were the most abundant birds breeding on the burn. Rodents remained virtually absent from burned plots.

Results of this 1987 fire, and others we have studied previously at the Research Ranch (Bock et al. 1976; Bock and Bock 1978, 1987), suggest the following about fire and wildlife in grasslands on the sanctuary:

1. Fire has been a natural and powerful evolutionary force shaping this ecosystem. Most species are fire tolerant; some are fire-dependent.
2. Livestock grazing interrupts the fire regime in this ecosys-

[1]Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).

[2]Department of Environmental, Population, and Organismic Biology, University of Colorado, Boulder, CO 80309, and Appleton-Whittell Research Ranch Sanctuary, National Audubon Society, P.O. Box 44, Elgin, AZ 85611.

tem, making it difficult to evaluate the natural effects of fire on vegetation and wildlife, or even to use fire as an effective management tool.

3. The sanctuary's grasslands recover quickly from the effects of fire, with grasses and herbs returning to pre-burn conditions in three to four years.

4. Prescribed fire should be used when possible to create rather fine-scale mosaics of stands at various stages of post-fire ecological succession. Probably no site should be re-burned at least until its grasses and herbs have recovered to pre-burn conditions. Some wildlife species require heavy cover typical of areas protected from fire as well as grazing; others thrive on increased resources temporarily available following a recent burn; still other species need both sorts of habitats simultaneously.

A necessary first step in reviewing the effects of burning on wildlife across the full range of lowland southwestern ecosystems is to group the landscape into major habitat types. The excellent volume edited by D. Brown (1982) provides a logical starting point. However, this treatise divides the study area into more biotic communities than we can reasonably consider. Furthermore, as noted by Lowe and Brown (1982), many of these communities grade into one another both spatially and through time. Of particular relevance to the present review, fire may have played a major role in the past in maintaining many southwestern sites as grassland, where today they are dominated largely by desertscrub.

The landscape units we have chosen to examine correspond closely to those broad biogeographic provinces recognized by Lowe and Brown (1982, fig. 3). These are units with generally distinct floral and faunal histories (e.g. Axelrod 1985, Cronquist et al. 1972, Lowe and Brown 1982), among which fire may differ in historical importance and management applicability. These habitat associations are:

1. Great Basin shrubsteppe,

2. Interior chaparral,

3. Madrean evergreen woodland,

4. Chihuahuan shrubsteppe (including Chihuahuan desertscrub, semidesert grassland, and plains grassland), and

5. Sonoran and Mojave desertscrub.

For each of these landscape units we shall review 1) something of their distribution, composition, and history, especially as relates to fire, 2) known effects of fire on vegetation, and 3) wildlife responses to fire, based upon actual studies, or as can be projected from what is known of the species' habitat requirements. Severson and Rinne (this volume) consider similar information about fire in higher-elevation southwestern forests and woodlands.

Great Basin Shrubsteppe

The most common vegetation throughout the Intermountain West is Great Basin shrubsteppe, with its most conspicuous shrub being big sagebrush (*Artemisia tridentata* L.). Particularly in the northern Great Basin, an understory of cool-season perennial grasses is (or was) important.

Dominant species are in the genera *Agropyron*, *Festuca*, and *Stipa* (Wright et al. 1979). The exotic annual cheatgrass (*Bromus tectorum* L.) has spread widely within historic times (Mack 1981), as the native perennials were damaged or destroyed by livestock.

Sagebrush communities in the southern Great Basin may never have had substantial grass cover, even before the introduction of livestock (Turner 1982a, Vale 1975). Yet it is very difficult to defend reconstructions of prehistoric conditions based on first written descriptions of this area, because these followed introduction of the horse. Even in the north, long term protection from grazing need not result in declines in sagebrush cover or increases in grass (Anderson 1986), though we believe such was the condition of these ranges prehistorically.

Boysag Point is a relatively isolated 70-acre tract on the north rim of the Grand Canyon, dominated today by big sagebrush (Schmutz et al. 1967). Perennial grass cover was 8.6% here, compared to 1.3% on an adjacent "mainland" site with a long history of livestock grazing. Warm-season grasses such as black grama (*Bouteloua eriopoda* Torr.) and galleta [*Hilaria jamesii* (Torr.) Benth.] were present, along with cool-season *Poa* and *Stipa* species. These data suggest strongly that grass cover also was important in southern Great Basin shrubsteppe before the depredations of domestic grazers, though likely not to the same extent as farther north.

Humphrey (1974) felt that Great Basin shrubsteppe burned more frequently than any other North American desert. Furthermore, because big sagebrush does not re-sprout following fire, burns in this habitat have the potential to drastically change the structure and composition of these plant communities. However, this presumes the presence of sufficient grass cover to carry fire from one shrub to the next, as well as to provide a source of grass propagules to colonize and spread following the burn.

Whatever the prehistoric importance of fire in Great Basin shrubsteppe, the Southwest can figure only

peripherally in this debate, since the habitat itself is only marginally present in the region. Sagebrush-dominated shrubsteppe occurs in the Arizona Strip north of the Grand Canyon, and in the Painted Desert region of northeastern Arizona and northwestern New Mexico (Turner 1982a).

Fire and Vegetation

Fire has been little-studied in Great Basin shrubsteppe of the Southwest. The Bureau of Land Management has been experimenting with prescribed burns in its Arizona Strip District (F. L. Leavitt, pers. comm.). Despite an initial scarcity of grass cover, a very high percent kill of sagebrush was achieved, with little shrub reinvasion in the first few post-fire years. Burns were seeded by broadcast or drilling, with an especially good establishment of sand dropseed [*Sporobolus cryptandra* (Torr.) Gray] and sideoats grama [*Bouteloua curtipendula* (Michx.) Torr.]. These managers are to be commended for using native grasses in their re-seeding program.

Elsewhere in the Great Basin, fire has been used extensively as a land management tool. Burning generally has proven the most economical and effective means of reducing big sagebrush (e.g. Wambolt and Payne 1986). Native grasses and forbs can be damaged by hot or too frequent fires, but they are generally tolerant of spring and fall burns. Reduced shrub cover at least potentially provides the opportunity for increases in grass and forb species (see Wright et al., 1979, for details).

Fire and Wildlife

We are unaware of any studies specifically designed to test the effects of burning on wildlife in Arizona or New Mexico sagebrush habitats. However, BLM personnel have seen large influxes of mule deer following prescribed burning and seeding in the Arizona Strip. Presumably this resulted from "the large increase in forbs and grasses and the resprouting of browse species" (F. L. Leavitt, pers. comm.). Hobbs and Spowart (1984) found that prescribed burning elevated the nutritional quality of mule deer and mountain sheep *(Ovis canadensis)* diets in winter, but not in spring, in Colorado sagebrush-grassland. These changes did not result from increased nutritional value of individual plants, but rather from increased availability of grasses in the burned areas.

While burning may cause temporary increases in certain high-quality grass and herb forage, it is important to remember that many Great Basin shrub species provide essential winter deer browse. Most of these species, such as sagebrush and antelope bitterbrush [*Purshia tridentata* (Pursch.) DC.], are damaged or eliminated by fire (Severson and Medina 1983). Pronghorn antelope similarly may benefit from increased forb cover following fire, but only if sufficient shrub browse is spared for the winter (Wright and Bailey 1982). A mosaic of burned and unburned stands probably provides the best overall big game habitat, although this needs to be tested specifically for southwestern sagebrush stands.

Because burning potentially can convert Great Basin shrubsteppe into pure and relatively persistent grassland, it can have a dramatic impact on those wildlife species sensitive to structural habitat features. This has been particularly well-studied with regard to songbirds, many of which require shrubs for nesting or as song perches. Although Great Basin shrubsteppe birds have rather wide ecological tolerances (Wiens and Rotenberry 1981), nevertheless certain species will be drastically affected by changes in shrub and/or grass cover. Sage thrashers *(Oreoscoptes montanus)* and sage sparrows *(Amphispiza belli)* would be eliminated by a complete sagebrush kill, whereas horned larks *(Eremophila alpestris)* and vesper sparrows *(Pooecetes gramineus)* are likely to increase (Peterson and Best 1988). A fire in ungrazed sage-grassland in southcentral Montana completely eliminated all sagebrush, including its stems and branches. Only western meadowlarks *(Sturnella neglecta)* nested on the burn in the first three years following the fire (Bock and Bock 1987), whereas adjacent unburned sites supported meadowlarks plus four additional species: lark sparrow *(Chondestes grammacus)*, lark bunting *(Calamospiza melanocorys)*, grasshopper sparrow, and Brewer's sparrow *(Spizella breweri)*.

Conclusions

Prescription burning can have beneficial impacts on wildlife in Great Basin shrubsteppe, as long as it is used to create rather fine-scale mosaics of stands in various stages of post-fire succession. Fire can increase the quality and quantity of grasses and forbs, but it also reduces cover and browse. We suspect that mosaics of burned and unburned shrubsteppe represent the condition of most Great Basin lowlands prehistorically, when the native grassland understory was relatively intact.

Interior Chaparral

There are no North American ecosystems better characterized as evolving under the influence of fire than chaparral (e.g., Conrad and Oechel 1982, Sweeney 1956). Interior chaparral is best developed in a band south of the Mogollon Rim, from northwestern Arizona into extreme southwestern New Mexico (Carmichael et al. 1978, Pase and Brown 1982). Shrubs predominate on unburned sites, often forming a nearly complete canopy. Lightning-caused wildfires are common (Bolander 1982), and top-kill of shrubs can be very high, but the shrub species

are adapted to recover and recolonize quickly. Dominant species such as shrub live oak (*Quercus turbinella* Greene) and mountain mahogany (*Cercocarpus* spp.) re-sprout from root crowns (Pase and Granfelt 1977), while others such as manzanita (*Arctostaphylos* spp.) and ceanothus (*Ceanothus* spp.) reproduce from large and persistent seed banks that germinate following fire. Grasses and herbs are scarce where shrubs predominate, but they can become temporarily abundant after fires (Pase and Brown 1982).

Historically, management efforts in Arizona chaparral have been directed at reducing shrub cover and increasing grasses, principally to improve water yield and livestock production (Brown et al. 1974, Hibbert et al. 1974). Prescribed burning has been an important tool in these chaparral "conversion" efforts. However, the shrubs recover within 5 to 10 years following burning (Pase and Granfelt 1977). Therefore, other control methods such as root plowing and herbicide treatment usually also have been used (Baldwin 1968, Hibbert et al. 1974), along with seeding of African lovegrasses (*Eragrostis* spp.).

Fire and Vegetation

Fire in Arizona chaparral appears not to result in long-term increases in grass and herb cover, except when coupled with repeated re-burns, or when followed by application of herbicides and seeding with exotic grasses (Pase and Pond 1964, Pase and Knipe 1977). Unseeded chaparral stands recover in 5 to 10 years, though they may not carry fire for as long as 20 years (Cable 1957, Hibbert et al. 1974, Pase and Pond 1964). Pase and Knipe (1977) studied the effect of a winter prescribed burn on a converted chaparral watershed in the Tonto National Forest. The area had been previously burned, treated with herbicide, and seeded with a mixture of African lovegrasses. The winter fire did not reduce subsequent production by the exotic grasses, but it did increase abundance of a dominant annual herb, *Ipomaea coccinea* L. Because of the complex pre-burn unnatural treatment of the area, and the fact that a winter burn is outside the natural fire season, it is difficult to determine what this study tells us about the natural role of fire in interior chaparral. Mayland (1967) found increased nitrogen availability in soils under scrub oak and mountain mahogany chaparral that was treated with herbicide and then control-burned in the fall.

Fire and Wildlife

There is general agreement that prescribed fire will be most beneficial to wildlife in interior chaparral if it is used to create relatively small openings, especially in areas with heavy cover of the less palatable shrubs (Brown et al. 1974, Hibbert et al. 1974, Severson and Medina 1983). Chemical treatment has the undesirable effect of eliminating palatable and well as unpalatable browse (Severson and Medina 1983). At least in southeastern Arizona, stands dominated by exotic African lovegrasses support a greatly diminished variety and abundance of wildlife compared to native grasslands (Bock et al. 1986). Burning outside the natural fire season could select against native plants and animals that have lived in this region since the last glaciation.

Few studies have considered the responses of individual wildlife species to burning of interior chaparral. More such work has been conducted in California chaparral, and we have attempted to extrapolate from these results to the southwestern situation where it is appropriate.

Szaro (1981) compared bird populations in Arizona chaparral with those of adjacent grassland and riparian habitats created over the preceding 20 years by repeated burning, seeding with African lovegrasses, and herbicide applications. Because the riparian zone, created by increased water yield in the manipulated area, was small relative to grassland, the overall effect of the conversion was a substantial loss of breeding birds. Densities ranged from a high of 321 pairs per 40 ha in riparian, to 103 and 24 pairs in chaparral and grassland, respectively. Species most abundant in untreated chaparral included Gambel's quail (*Callipepla gambelii*), Bewick's wren (*Thyromanes bewickii*), bridled titmouse (*Parus wollweberi*), and rufous-sided towhee (*Pipilo erythrophthalmus*). Rock wren (*Salpinctes obsoletus*) and rufous-crowned sparrow (*Aimophila ruficeps*) were restricted to grassland. Eight species, including especially Bell's vireo (*Vireo belli*), yellow warbler (*Dendroica petechia*), and Scott's oriole (*Icterus parisorum*), were most abundant in the riparian site. Breeding birds may have been uncommon in the converted grassland because of the dominance of the African exotics (Bock et al. 1986).

Lawrence (1966) compared breeding birds of burned vs. unburned chaparral in the Sierra Nevada foothills in California. Total bird numbers were little-changed by the fire, but species' relative abundances were substantially affected through three post-fire years. Brushland species, such as the California quail (*Callipepla californica*), Bewick's wren, and brown towhee (*Pipilo fuscus*) declined, while grassland birds such as the mourning dove and western meadowlark increased.

We can find no data on rodent responses to fire in Arizona chaparral. Several studies in California brushlands show that fire has a dramatic, if short-term, effect on this group of mammals. Fires probably cause little direct rodent mortality, since most species survive in underground burrows. However, woodrats (*Neotoma* spp.) living in above-ground nests are killed in large numbers (Chew et al. 1959). Fires initially appear to

cause all rodents to abandon chaparral (Cook 1958, Wirtz 1982). However, recolonization is rapid, especially for those species preferring open or grassland habitats, such as voles (*Microtus* spp.), pocketmice (*Perognathus* spp.), and harvest mice (*Reithrodontomys* spp.). Shrub specialists such as *Peromyscus truei* and *P. californicus* were unable to occupy burned California chaparral, due to lack of suitable habitat (Lawrence 1966).

Prescribed burning potentially can improve interior chaparral for deer, if it results in mosaics of stands in various stages of post-fire re-growth (Hibbert et al. 1974, Severson and Medina 1983). Young shoots and herbs could provide higher quality forage, while older stands may provide some cover. Excessive brush control, whether through fire or chemical treatment, could reduce habitat suitability for either white-tailed deer *(Odocoileus virginianus)* or mule deer (McCulloch 1972).

Fire has been used to improve California chaparral for mule deer (Biswell 1969, Taber and Dasman 1958). In northern California, mule deer herds increased 300%, and animals were in better condition, following initiation of a prescribed burning program (Thornton 1982). Large wildfires were excluded, while controlled burns were conducted on 8 to 10 ha parcels planned for a 25 to 30-year rotation. Perennial grass fuelbreaks also were burned, on a planned 10-year rotation. Deer herd improvements were attributed to (1) creation of desirable cover/forage ratios, (2) increased nutrient quality of shrub sprouts for about three post-fire years, and (3) increased quantity of grass forage on the fuel break sites for at least three post-fire years.

Conclusions

From a wildlife standpoint we question the desirability of herbicide treatments, seeding with exotic grasses, or burning outside the natural fire season, in chaparral ecosystems. .All have the potential to reduce the wildlife value of such areas (to say nothing of their impacts on the native flora), and especially if applied on too broad a geographic scale. An exception might be to use chemical treatments to create firebreaks for management of small-scale prescribed burns. Such small burns, if repeated on perhaps a 10 to 20 year rotation, almost certainly will increase the abundance and variety of all kinds of wildlife in interior chaparral. Reseeding, if necessary and practical, should involve the use of indigenous plants.

Madrean Evergreen Woodland

This ecosystem type is centered in the Sierra Madre of Mexico, but it extends northward into the foothills and mountains of southeastern Arizona, southwestern New Mexico, and Trans-Pecos Texas (D. Brown 1982). At higher elevations it grades into pine-oak woodland, while at lower elevations it is open and savannah-like. This lower-elevation savannah, or encinal, is the focus of this review.

Dominant trees are oaks, most particularly Emory (*Quercus emoryi* Torr.), Arizona white (*Q. arizonica* Sarg.), Mexican blue (*Q. oblongifolia* Torr.), and gray (*Q. grisea* Liebm.). Alligator bark and one-seed juniper [*Juniperus deppeana* Steud and *J. monosperma* (Engelm.) Sarg.] and Mexican piñon (*Pinus cembroides* Zucc.) also are locally common (D. Brown 1982). Perennial grasses can be very well developed, especially on level, lowland, and ungrazed sites. Grass species diversity appears to be very high, some common species being sideoats grama, plains lovegrass (*Eragrostis intermedia* Hitch.), Texas bluestem (*Andropogon cirratus* Hack.), and muhlys (*Muhlenbergia* spp.).

There is much debate about the prehistoric role of fire in southwestern oak savannah (Hastings and Turner 1965, Severson and Medina 1983), and not much evidence upon which to rely. Bahre (1985) reports on 33 fires in southeastern Arizona that were described in local newspapers between 1859 and 1890; some of these certainly involved the encinal. Humphrey (1987) visited and photographed U.S.-Mexico border monuments in 1983-84, and then compared his photos with those taken in 1892-93. Many upper elevation evergreen woodland sites are more heavily wooded today than they were in the 1890's, which Humphrey attributes at least partially to fire suppression policies in place over the past 90 years. However, the lower elevation encinal sites appeared relatively little-changed.

Fire and Vegetation

Johnson et al. (1962) studied juniper and oak mortality following a hot June wildfire in southeastern Arizona. Mortality of Emory oak and Arizona white oak was 10% to 20% on the burn, but these results were difficult to interpret because of apparent drought-related oak mortality on the adjacent control area. Fire killed nearly 80% of one-seed juniper in all size classes, but only 32% of alligator junipers less than 3 inches in diameter, and only 23% of larger alligator junipers. Basal sprouting of burned alligator juniper was 42%, while only 10% of one-seed juniper sprouted after the fire.

Gawith (1987) studied the effects of a May prescription fire on alligator juniper invading grassland on Fort Huachuca. Fuel levels were high because the area had been ungrazed for about 30 years. Fire intensity ranged up to 2800 kilowatts per meter of fire front. Only 13% of burned trees died, but mortality was highest in small size classes. Gawith concluded that repeated burning would prevent alligator juniper invasion into ungrazed encinal grasslands.

Bock and Bock (1987) studied the effects of small (600 m^2) prescribed burns in oak savannah on the Appleton-Whittell Research Ranch, west of the Huachuca Mountains. Burns took place on 25 May 1984, when temperature was 33 C, relative humidity 17%, and winds generally calm. The fires were cool and of short duration. Scattered mature Emory and Arizona white oak were unaffected by burning, and oak seedling densities did not differ on burn vs. control plots through two post-fire growing seasons. Shrub densities were not affected, but heights of two species, *Mimosa biuncifera* Benth. and *M. dysocarpa* Benth., were significantly reduced on burned plots through two post-fire years. Perennial grass cover was reduced on burned plots by about 27% through one year, but this fire effect disappeared after two years. Herb cover nearly doubled on burned plots relative to controls in the first year, but again this difference disappeared by the end of the second post-fire year.

Our study plots were burned again by the extremely hot wildfire described in the introduction to this review. Our impression is that more woody plant mortality occurred as a result of this more intense fire, but such a conclusion must await data collection through more post-fire years.

Fire and Wildlife

Very little is known about wildlife responses to fire in low-elevation encinal. Barsch (1977; cited in Severson and Medina 1983) found that white-tailed deer benefitted from a wildfire that stimulated desirable browse in upper encinal. R. Brown (1982) found that Montezuma quail *(Cyrtonyx montezumae)* in southeastern Arizona are dependent upon dense perennial grasses as escape cover. These quail disappeared from heavily grazed sites, even though food supplies remained abundant. Presumably large-scale fires could have a similar negative impact on Montezuma quail.

We studied the responses of birds, rodents, and vegetation to a human caused February wildfire in open oak savannah on the Research Ranch, six years after livestock removal (Bock et al. 1976). Because no pre-burn data were collected, the legitimacy of our conclusions depends on the validity of comparisons between burned and adjacent unburned areas. There was no sign of tree mortality on this Research Ranch burn, although up to half the leaves on many oaks were scorched and killed. Grass cover was reduced through two post-fire summers, but there was little change in herb cover. Seed production was much higher on the burn in the first post-fire year.

Birds collectively were about 18% more abundant on the burn over 18 post-fire months. Two seed-eating species, mourning dove and chipping sparrow *(Spizella passerina)*, were largely responsible for this difference. The grasshopper sparrow is dependent on heavy grass cover in southeastern Arizona (Bock and Webb 1984), and it disappeared entirely from the burned area for the duration of the study. Rodents were about 40% less common on the burn than on the adjacent control, and no species was trapped significantly more often on the burn. White-throat woodrats *(Neotoma albigula)*, whose nests were burned away, and least cottonrats *(Sigmodon minimus)*, grazing rodents typical of dense grass, were virtually absent from the burn through two post-fire summers.

Conclusions

Much more research is needed into the responses of vegetation and wildlife to prescribed burning in lowland encinal. Managers planning to burn this habitat should try to notify the research community early enough to permit pre-burn sampling.

The impact of fire in encinal is likely to depend heavily on tree densities, ground fuel levels, and atmospheric conditions. Cool fires should kill few trees, but they may temporarily reduce grass cover, stimulate herbs, and increase seed production. Reduced ground cover should favor songbirds over rodents.

Sonoran Desertscrub

This diverse biome occupies most of lowland southwestern Arizona, grading to a small and uncertain degree into Mojave desertscrub in the northwestern part of the state (Humphrey 1974, Shreve 1942, Turner 1982b, Turner and Brown 1982). The vegetation is complex, both within and between localities. Creosote-bush [*Larrea tridentata* (DC) Coville] and white bur sage [*Ambrosia dumosa* (Gray) Payne] are dominant shrubs in lowland sites. Sonoran uplands are characterized by a wide variety of trees and large cacti, including ironwood (*Olneya tesuta* Gray), palo verde (*Cercidium* spp.), mesquites (*Prosopis* spp.), and saguaro [*Carnegia gigantea* (Engelm.) Britt. & Rose].

There is much debate about whether a true "desert grassland" is (or ever was) an important part of the Sonoran Desert in Arizona. Even more controversial is the role that fire, lightning or human-caused, may have played in reducing woody vegetation or maintaining such a grassland prior to European colonization (Bahre 1985, Dobyns 1981, Hastings and Turner 1965). Certainly it is gone today, except for swales and lowlands occupied by big galleta [*Hilaria rigida* (Thurb.) Benth.] or tobosagrass [*H. mutica* (Buckl.) Benth.].

Humphrey (1974, 1987) considered fires to be rare and generally unimportant in the Sonoran and Mojave deserts, except in *Hilaria* stands or in places where the desert intergrades with semidesert grassland. Wright and Bailey (1982) do not

even include a chapter on the Mojave or Sonoran deserts in their general treatise on fire ecology. Nevertheless, fires do occur in these deserts today, following years when exceptional rains produce a heavy ground cover of annual herbs and grasses (Cave and Patten 1984, McLaughlin and Bowers 1982, Pase and Granfelt 1977, Roundy 1986). Many of these annuals are introduced, however, so it remains unclear how often native annuals could have carried fire (Rogers and Steele 1980, Roundy 1986).

In the absence of dendrochronological data, Rogers and Steele (1980) attempted to use degree of fire adaptation in perennial plants as evidence for historical fire frequency in the Sonoran Desert. They concluded that "...positive adaptations are common, but are weakly developed" (p. 15), that post-fire recovery time is very long (up to 20 years), and that fire management should be generally conservative.

Fire and Vegetation

Wildfires and controlled burns in upland Sonoran Desert sites have caused substantial mortality of woody plants and cacti. Fire reduced density and cover of perennial plants by 91% and 84%, respectively, on the Granite Burn near Florence, Arizona (McLaughlin and Bowers 1982). Mortality of bur sage was 92%, creosote-bush 61%, and palo verde 63%. Mortality of saguaros was 31% after 19 post-fire months, and consisted mostly of individuals $\leq$2 m tall; but saguaro death on this same area rose to 68% after 54 months, and included many large plants (Rogers 1985).

It is not clear whether Sonoran Desert fires necessarily even increase ground cover. Cave and Patten (1984) found that fire reduced annual plant density but increased biomass. This seeming paradox is largely explained by fire-caused reductions in dense red brome (*Bromus rubens* L.), coupled with increased biomass of schismus grass (*Schismus arabicus* Nees.). Both these annual grasses are exotic. Herbs, native and introduced, showed mixed responses to the fires.

Fire and Wildlife

We have found no data on wildlife responses to fire in Sonoran desertscrub. Some projections can be made, based upon general knowledge about the habitat and food requirements of particular groups. The herpetofauna and avifauna of Sonoran Desert uplands are very rich (Phillips et al. 1964, Turner and Brown 1982, Tweit and Tweit 1986). Because most species depend upon the trees, shrubs, and cacti at least for cover and breeding sites, fire-caused mortality of these plants would have a highly negative impact on these groups. As an example, some of the best-known and most characteristic Sonoran Desert birds nest in cavities in saguaros. Fire-kill of this cactus would do great harm to these species, including flicker *(Colaptes auratus)*, Gila woodpecker *(Melanerpes uropygialis)*, brown-crested flycatcher *(Myiarchus tyrannulus)*, and elf owl *(Micrathene whitneyi)*. Many rodents would respond positively to increases in seeds produced by the desert annuals, but at least today these events are triggered largely by rainfall rather than fire. Mule deer occupy Sonoran Desert (Ordway and Krausman 1986), but seem to prefer areas with substantial cover. Desert bighorn *(Ovis canadensis nelsoni)* occupy only a small part of their original southwestern range. Bighorn sheep generally are thought to prefer grasses, and fire might be used on a limited scale to increase their forage (Monson and Sumner 1980).

Conclusions

Fire historically may have helped maintain an herb/grassland in parts of the Sonoran Desert now dominated almost exclusively by shrubs, trees, and cacti. Most evidence suggests that fires under present conditions are highly destructive of this perennial vegetation, without necessarily increasing cover of herbs and grasses, annual or perennial. More research is needed on this subject, including measuring responses of wildlife to wildfire. In the meantime, we can see little benefit and much potential harm coming to wildlife as a result of fire, wild or prescribed, in Sonoran desertscrub.

Chihuahuan Shrubsteppe

The biome we choose to call Chihuahuan shrubsteppe includes three community types: Chihuahuan desertscrub, semidesert grassland, and plains grassland (D. Brown 1982). There are three reasons why we have combined them for the present review. First, as noted by Lowe and Brown (1982:16), the boundaries between them are difficult to discern, at least under present circumstances. Second, these communities have a common, essentially Chihuahuan, ancestry (Axelrod 1985). Finally, fire and fire-exclusion have strongly influenced their composition and distribution.

Before recent desertification, true Chihuahuan desertscrub occurred largely in Mexico, extending into the U.S. only in southwestern New Mexico, up the Rio Grande Valley, and into a small part of extreme southeastern Arizona (D. Brown 1982, Schmidt 1979). Lowland plains, usually of limestone origin, were dominated by creosote-bush, tarbush (*Flourensia cernua* DC), whitethorn acacia (*Acacia constricta* Benth.), or honey mesquite [*Prosopis juliflora* var. *glandulosa* (Torrey) Cockerell.]. Higher slopes supported mixtures of succulents, especially of *Agave* and *Yucca*.

Semidesert grasslands occupied lowland sites in southern New Mex-

ico and southeastern Arizona, but above Chihuahuan desertscrub (D. Brown 1982). Tobosagrass and black grama are (or were) predominant, along with a variety of other perennial grasses. Shrubs in such genera as *Mimosa*, *Acacia*, and *Prosopis* always have been part of the semidesert grassland, but prehistorically these were less abundant or widespread than is the case today.

Plains grassland characterizes the lowlands of northern and eastern New Mexico, and perhaps elevations intermediate between those of semidesert grassland and encinal in southeastern Arizona (D. Brown 1982). Originally this landscape was nearly a pure grassland, dominated by blue grama [*Bouteloua gracilis* (H.B.K.) Lag.], buffalo-grass [*Buchloe dactyloides* (Nutt.) Engelm.], and other perennials.

Drought, grazing, and fire are major factors affecting the biogeography, ecology, and evolution of Chihuahuan shrubsteppe. So strong are the interactions among these factors that it is impossible to consider fire, from either an historical or management perspective, except in the context of the other two.

Chihuahuan desertscrub has invaded many areas of New Mexico that formerly were grassland (Buffington and Herbel 1965). Much of this can be attributed to the devastating impacts of livestock grazing. However, subsequent livestock exclosure has not stopped the loss of black grama at the Jornada Experimental Range in southern New Mexico, where mesquite is invading and increasing (Hennessy et al. 1983). This continued loss of black grama may be due in part to long-term increases in aridity (Neilson 1986).

Humphrey (1987) documented significant loss of grass cover at monuments along the New Mexico-Chihuahua border between 1893 and 1983. He attributed these to (p.429) "...a slight but consistent climatic trend toward increasing aridity combined with long-exerted grazing pressures." Fires may once have been locally important in controlling Chihuahuan Desert shrubs, but they lost all influence once overgrazing destroyed the fragile grasslands.

Vegetation changes have been well-documented at the semidesert grassland Santa Rita Experimental Range in southern Arizona (Martin 1986). Once relatively pure stands of black grama and other native grasses predominated. Today, burro weed [*Haplopappus tenuisectus* (Greene) Blake], cholla (*Opuntia fulgida* Engelm.), and mesquite [*Prosopis juliflora* (Swartz) DC] are abundant. Native grasses have largely been replaced by the African exotic lovegrass, *Eragrostis lehmanniana* Nees. These changes can be attributed to the combined effects of continued livestock grazing, early fire exclusion, and introduction of the exotic. At least on the Research Ranch, stands of African lovegrasses are biologically sterile compared to adjacent stands of native vegetation (Bock et al. 1986).

Humphrey (1974, 1987) believes that fire played a major historical role in controlling woody plants in semidesert grassland. Fires were equally common and important in plains grassland (Jackson 1965, Vogl 1974), and shrub invasions have occurred here as well following overgrazing and fire suppression. Paradoxically, the dominance of plains grassland in higher elevation sites in southeastern Arizona may be an artifact of livestock grazing. Bison *(Bison bison)* have not occurred in southeastern Arizona for at least the past 10,000 to 12,000 years (McDonald 1981; P.S. Martin, pers. comm.), so that cattle were a truly exotic force when they were introduced. Blue grama is perhaps the quintessential plains grassland species, and it dominates grazed areas today in parts of southeastern Arizona (Bock et al. 1984, Bock and Bock 1986). Yet on livestock exclosures, taller grasses such as plains lovegrass, wolftail (*Lycurus phleoides* H.B.K.), and certain bluestems (*Andropogon* spp.) become common, eventually at the expense of blue grama. However such grasslands are to be classified, we believe they too evolved with and were maintained by fire prehistorically.

Fire and Vegetation

The impact of fire on Chihuahuan shrubsteppe vegetation has been well studied and thoroughly reviewed (Martin 1975, Pase and Granfelt 1977, Wright 1980). While it may have broader wildlife applications, the objective of most prescribed burning has been to reduce woody plants and stimulate livestock forage. The efficacy of these attempts seems rather site-specific, depending upon such things as precipitation patterns, stand species composition, and range condition.

Black grama can be severely damaged by fire in lowland sites (Cable 1965), though this grass also has been stimulated by burns (e.g. Ahlstrand 1982). Most plains and semidesert grasses recover from fire in one to three years, and some may be encouraged by it (Bock and Bock 1987, Wright 1980). Tobosagrass may either be stimulated or temporarily reduced by fire, depending upon precipitation following the burn (Neuenschwander et al. 1978).

Shrubs such as burro weed and creosote-bush are killed by prescribed burning, if fuel levels are sufficient to carry fire (Cable 1973). Small mesquite are killed by fire, but individuals over 5 cm diameter are relatively invulnerable. Frequent fire in undisturbed grasslands historically may have kept large areas free of mesquite. Yet recent invasions into disturbed sites can be reversed only by more drastic measures such as root plowing and chemical treatment (Martin 1975), or perhaps by dramatic natural events such as rapidly repeated wildfires, extremely low winter temperatures, or a combination of both.

Wright (1980) concluded that fire should not be used on black grama ranges, but that it can be an effective tool for reducing certain shrubs on mixed grama areas and in tobosagrass. However, effectiveness of prescribed burning depends upon good grass cover.

Relatively little is known about the response of herbs to fire in Chihuahuan shrubsteppe (Wright 1980), yet this component may be a critical one for certain wildlife. Relatively cool prescribed burns in an ungrazed *Mimosa-Bouteloua-Eragrostis intermedia* community on the Research Ranch had little impact upon grasses or forbs (Bock and Bock 1987); however, hot wildfires, such as the one described in the introduction to this review, stimulated herb production (see also Bock et al. 1976).

Fire and Wildlife

Compared to vegetation and livestock, the responses of wildlife to burning Chihuahuan shrubsteppe have been surprisingly little-studied. Much of what we can say about wildlife here must be based upon general knowledge of species' food and habitat requirements.

Wildfires at the Research Ranch have had mixed impacts upon passerine birds (Bock et al. 1976, Bock and Bock 1988). Common winter species such as vesper sparrows and chipping sparrows concentrate on fresh burns in very large numbers, probably because of increased seed production. Species preferring bare ground, such as the horned lark and lark sparrow, breed in high numbers on either grazed or recently burned sites. Three sparrows characteristically nest in unburned and ungrazed grasslands at the Research Ranch: Cassin's *(Aimophila cassinii)*, Botteri's *(A. botterii)*, and grasshopper. These species are scarce or absent on grazed sites (Bock and Bock 1988), and they also avoid burns for one or two post-fire growing seasons.

Fire-caused reductions in woody plants are not necessarily beneficial to song or gamebirds. Bock and Webb (1984) found that while Cassin's and grasshopper sparrows selected sites with heavy grass cover, they also were associated with scattered mesquite and low shrubs used as song-perches. Renwald (1978) found that scattered mesquite and lotebush [*Zisiphus obtusifolia* (Hooker) Gray] were an important habitat component for songbirds nesting in tobosagrass in central Texas. Maurer (1985) found that eleven songbird species nested in greater numbers on parts of the Santa Rita Experimental Range with numerous mesquite, while ten different species preferred relatively open grassland.

Mourning doves concentrate in very large numbers on burned portions of the Research Ranch in the first fall after a wildfire (Bock and Bock 1988). Presumably this is because of increased forb seed production (Best and Smartt 1986, Bock et al. 1976). However, Germano et al. (1983) found that mourning dove, scaled quail *(Callipepla squamata)*, and Gambel's quail all preferred grassland areas with partial mesquite cover more than areas completely cleared of mesquite (see also Ault and Stormer 1983). A mosaic of sites with varying mesquite densities also would maintain high reptilian diversity (Germano and Hungerford 1981).

Mule and white-tailed deer require some woody cover (e.g. Quinton et al. 1979, Steuter and Wright 1980), but could perhaps benefit from increased grasses and high-quality browse sprouts following fire in semidesert and plains grassland (Anthony and Smith 1977, Severson and Medina 1983). Forbs were a very important component of the winter diet of mule deer and pronghorn in New Mexico (Stephenson et al. 1985). More research is needed to determine if these ungulates would respond positively to fire-caused forb increases in Chihuahuan shrubsteppe.

Conclusions

As with Great Basin shrubsteppe and interior chaparral, prescribed fire will benefit wildlife in Chihuahuan shrubsteppe if it is used to create mosaics. Fire can stimulate herb, seed, and perhaps grass production. Scattered shrubs and mesquite are likely to enhance the wildlife value of most areas, compared either to dense stands of woody vegetation or to pure grasslands. Ground cover should return to pre-burn conditions in about three growing seasons.

The most serious negative impacts on wildlife in semi-arid grasslands are from livestock grazing and the spread of exotic grasses. Fire cannot solve either of these problems. However, prescribed burning can be an integral and natural part of wildlife management in these ecosystems, especially if they can be restored to something resembling their prehistoric condition.

Miscellaneous Habitats

Riparian Woodland

Southwestern riparian woodlands are major centers of wildlife diversity, especially for birds (Johnson and Jones 1977, Johnson et al. 1985). In central Arizona, a complex history of wildfire, chemical treatments, and prescribed burning in interior chaparral increased runoff and stimulated growth of riparian trees (Szaro 1981). However, fire is difficult to manage and potentially very destructive in established riparian woodlands. At the Research Ranch, wildfires have killed mature cottonwood (*Populus fremontii* S. Wats.), sycamore (*Platanus wrightii* S. Wats.), velvet ash (*Fraxinus velutinus* Torrey) and Arizona walnut (*Juglans major* Heller). We do not recommend prescribed

burning in these limited habitats (see also Severson and Rinne, this volume).

Sonoran Savanna Grassland

This habitat once occupied lowland plains and bottomlands in the Altar and Santa Cruz valleys of southern Arizona (D. Brown 1982). Dominant grasses were *Bouteloua* and *Aristida* species, while mesquite was an important, if scattered, overstory tree. Heavy grazing and fire suppression caused native grasses to decline and shrubs to increase within the past century. In addition, Lehmann's lovegrass has spread into the area.

The masked bobwhite quail *(Colinus virginianus ridgwayi)* once occupied Sonoran savannah grassland in Arizona and Mexico. This endangered subspecies became extinct in Arizona when its habitat was degraded to desertscrub (Goodwin and Hungerford 1977, Stromberg et al. 1986). The Buenos Aires Wildlife Refuge was established in the Altar Valley as a site for re-establishment of masked bobwhite in Arizona. Prescribed burning may prove to be an important tool in this recovery effort (W. Shifflett, pers. comm.). Recent wildfires and controlled burns have significantly reduced burro weed and snake weed [*Gutierrezia sarothrae* (Pursh.) Britton], set back mesquite, and possibly stimulated native perennials relative to Lehmann's lovegrass.

Sacaton Floodplains

Big sacaton (*Sporobolus wrightii* Munro.) is a coarse, tall, perennial bunchgrass that forms nearly monotypic stands in broad floodplains of southeastern Arizona and southwestern New Mexico. These stands have been severely degraded historically by de-watering, channelization, and erosion (Cox et al. 1983). Sacaton frequently is burned to increase its value as livestock forage (Cox 1988). Spring and fall burning have long-term negative impacts, but sacaton burned in the natural mid-summer season recovers to pre-fire standing crop in three years (Cox and Morton 1986).

Wildfires in ungrazed sacaton stands on the Research Ranch have dramatic but short-term wildlife consequences (Bock and Bock 1978, 1979). Collared peccary *(Dicotyles tajacu)* use mature stands for cover, while many important food plants grow on recent burns. Yellowthroats *(Geothlypis trichas)*, blue grosbeaks *(Guiraca caerulea)*, and Botteri's sparrows commonly nest in mature sacaton. However, doves, quail, and many wintering sparrows are attracted to large seed crops produced by forbs in recently burned stands. Grazing cottonrats are very abundant in unburned areas, whereas burns can support large populations of seed-eating rodents such as the hispid pocket mouse *(Perognathus hispidus)* and Merriam's kangaroo rat *(Dipodomys merriami)*.

Undisturbed sacaton appears to recover from fire in two to three years. A mosaic of stands in various stages of post-fire re-growth would be maximally beneficial to wildlife.

Management Implications And Recommendations

1. From a wildlife perspective, we believe the goal of land management should be to maintain or return ecosystems to their natural state, because such habitats will support the greatest diversity of native plants and animals. By natural state we mean the general condition in which they existed since the last glaciation but prior to European colonization. This goal should take precedence over management of a habitat for one particular species, except under very unusual and restricted circumstances. Most particularly, this overall goal may be at odds with the objective of increasing forage for livestock Prescribed fire should be used when it is a tool for returning ecosystems to their natural state. We recognize this may be a problem (1) when it is not clear what was the natural state of a particular system, or (2) when historical habitat modifications make application of fire difficult, or (3) when such ecosystem restoration is in conflict with other land uses.

   Fire is a natural ecological and evolutionary force in many southwestern lowland habitats, and it is one to which most native plants and animals are variously adapted. Fires kill little wildlife directly (Bendell 1974,Wright and Bailey 1982). Burning in the natural fire season is less likely to damage vegetation than burning out of season (e.g. Cox and Morton 1986).

2. There is universal testimony to the value of using fire to create mosaics for the benefit of wildlife (e.g., Lyon et al. 1978; Pase and Granfelt 1977; Severson and Medina 1983; Severson and Rinne, this volume). Unburned stands provide essential cover, while fires can stimulate browse, herbs, and grasses. Since many species need both unburned cover and fire stimulated food, mosaics should be on an appropriately small geographic scale.

3. Based upon available information, fire is likely to be

more destructive than beneficial to wildlife and wildlife habitat in (1) Sonoran and Mojave desertscrub, (2) black grama ranges in the lower parts of the Chihuahuan shrubsteppe, and (3) riparian woodlands. More research is needed into the effects of fire on wildlife in Madrean evergreen woodland.

4. Prescribed fire is a potentially powerful wildlife management tool in (1) Great Basin shrubsteppe, (2) interior chaparral, (3) semidesert and plains grassland portions of Chihuahuan shrubsteppe, (4) Sonoran savanna grassland, and (5) sacaton floodplains. In all cases, however, fire frequency should not exceed the time required for post-burn recovery of herbs and grasses. Furthermore, the landscape mosaic always should include cover provided by shrubs and trees, where they are a natural component of the ecosystem.

5. Seeding burned areas with exotic grasses is highly undesirable, since stands of these grasses are sterile places for wildlife compared to mixtures of native species.

6. A major impediment to using fire creatively in southwestern lowlands is habitat alteration by domestic grazers. Reduced fuel loads prevent fires of a natural intensity. Fire-tolerant woody vegetation has become too well established, often crowding out grasses and herbs.

7. There is a great need for further research on the specific effects of prescribed burning on wildlife in southwestern lowlands. Fire-ecological experiments must be doubly-controlled; that is, burn and control study plots must be monitored both before and for 3-4 years after a fire. In addition to population density, it may also be important to measure species' survivorship and reproductive success on burned vs. unburned sites (Johnson and Temple 1986, Van Horne 1983).

Given the present emphasis on the value of field experiments in ecology in general, it should be possible to attract more university-based researchers, especially students, to study fire. Burning makes a dramatic and often spectacular ecological experiment. All that should be necessary on the part of managers is to contact researchers one or two years in advance, and then to burn where and when planned.

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Effects of Fire on the Soil Resource in Arizona Chaparral[1]

Leonard F. DeBano[2]

Abstract.—Both prescribed burns and wildfires consume aboveground biomass and litter and produce nutrient transformations that affect postfire management. Substantial quantities of important plant nutrients, like nitrogen (N) and phosphorus (P), are lost directly during combustion. Highly available nutrients released during a fire are deposited on the soil surface where they may be immobilized, or lost by erosion. Effect of available nutrients on postfire fertility is discussed. Information on the effect of fire on specific physical, chemical, and biological soil properties is used as a basis for discussing short- and long-term consequences of postfire rehabilitation treatments.

Chaparral covers 1.3-1.5 million hectares as a discontinuous band of vegetation extending across Arizona in a northwest to southeast direction (Hibbert et al. 1974). It is an important vegetation type on both the Prescott and Tonto National Forests and is present, but to a lesser extent, on the Coronado and Kaibab National Forests in Arizona. Wild and prescribed fires occur frequently throughout Arizona chaparral. Fire danger is most severe in spring and early summer until summer rains start, and then again during late fall after the summer monsoon season has ended. Prescribed burning can be done in this type throughout the year, although most burning is done during periods having less severe burning conditions. Because both wild and prescribed fires occur frequently throughout chaparral, land managers are continually asked to assess fire effects on different resources, including soils, while developing burning prescriptions or planning postfire rehabilitation treatments. The objectives of this paper are to (1) describe the chaparral resource, (2) discuss fire behavior in chaparral, (3) outline a general approach for assessing fire effects on chaparral soils, (4) present a detailed summary of the effects of fire on soil properties that can be expected during prescribed fires, (5) develop an example illustrating a procedure that can be used for assessing fire effects on chaparral soils, and (6) address some concerns arising as part of postfire management of burned areas. This paper focuses mainly on Arizona chaparral although applicable research findings on chaparral soils in California are used when needed to fill knowledge gaps. Detailed comparisons of fire effects and postfire rehabilitation between Arizona and California chaparral are presented elsewhere (DeBano, in press).

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[2]Leonard F. DeBano, Principal Soil Scientist, with Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture, Tempe, Arizona.

The Chaparral Resource

Vegetation and Evolution

The chaparral plant community is characterized by moderate- to deep-rooted evergreen sclerophyllous shrubs. The term "chaparral" is derived from the Spanish word "chapparo" referring to dwarf evergreen oaks (Shantz 1947). These shrubs reach their best development on deep soils or on deeply weathered or broken rock mantles. Arizona chaparral has been classified into several climax vegetation associations: mountainmahogany-mixed shrub, shrub live oak-hairy mountainmahogany *(Quercus turbinella-Cercocarpus breviflorus)*, shrub live oak-birchleaf mountainmahogany *(C. betuloides)*, shrub live oak-mixed shrub, Arizona cypress *(Cupressus arizonica)*-shrub live oak, and Arizona oak-yellowleaf silktassel-Emory oak *(Quercus arizonica-Garrya flavescens-Quercus emoryi)* (Carmichael et al. 1978). Most chaparral shrub species are prolific crown and/or root sprouters and produce few seedlings (Pase 1969); however, the few nonsprouters produce abundant seed. Sprouting species live to a considerable age; Pond (1971) reported little change in individual plants after 47 years.

Arizona chaparral vegetation originated from Madro-Tertiary geoflora during the Cenozoic era (Axelrod 1958) and evolved into its present floristics during the mid-Pliocene Epoch in response to major topographic-climatic changes, which produced its present climate. Arizona chaparral receives about 400-600 mm precipitation annually, distributed bimodally with approximately 55% during the winter from November through April, and the remaining 45% during summer convection storms in July through September (Hibbert et al. 1974). This climatic regime favors a spring and summer growing season for many of the important shrubs.

The adaptation of chaparral vegetation to fire has influenced the evolution of its species since the Miocene

Era (Axelrod 1958). As a result of natural selection, adaptations have developed that prevent the elimination of chaparral species by fire. These adaptations include (1) seed production at an early age; (2) fire-resistant seeds; (3) seeds depending on fire for germination; (4) production of seeds in such large numbers that some survive even though the heat destroys many; and (5) sprouting from latent buds mainly belowground (Hanes 1970). The shrub species in this type are well adapted to drought because their well-developed root systems can exploit large volumes of soil for water and nutrients (Davis and Pase 1977, Hellmers et al. 1955b), allowing the stands to regenerate rapidly following fire (Pase and Lindenmuth 1971, Pase and Pond 1964, Pond and Cable 1962). Favorable weather conditions following fire germinate seeds of nonsprouting shrubs, such as ceanothus (*Ceanothus* spp.)and manzanita (*Arctostaphylos* spp.), which, under natural conditions, do not germinate until they are scarified by heat (Pase 1965). Rates of recovery of individual shrub species after fire are dependent on their sprouting ability, ability to establish seedlings, their relative abundance in the prefire stand, and possibly the time of year of the burn (Cable 1975). In some cases, 11 years or more are required for shrubs to reach preburn cover conditions (Hibbert et al. 1974).

Soils and Geology

Soils supporting chaparral are typically deep, coarse textured, and poorly developed. Soil, as used in this discussion, includes all porous material (regolith) in which weathering and roots are active. The distinction between soil depth and solum depth (A and B horizons) is critical in this case, since most of the soil is in the C1 horizon. Consequently, soil surveys that describe soil as shallow generally pertain to solum depth, not depth of regolith. Usually, the A horizon is only a few inches thick, and the B horizon is commonly absent. Soil texture varies from cobbley and gravelly loamy sand to gravelly loam. The soils belong to several soil suborders, including Ochrepts, Orthents, Ustalfs, Ustolls, and Borolls.

Parent rock materials are made up of deeply weathered and fractured granite, schist, diabase, sandstone, shale, limestone, slate, gneiss, quartzite, and basalt. Granites are found on more than half the total chaparral area, with none of the other types making up more than 10%; schist-derived soils are probably the second most common. Rock types such as basalt, limestone, quartzite, shale, and slate, which weather to fine-textured and shallow regoliths, do not support extensive stands of chaparral, even though rainfall and elevation are often similar. Instead, these soils often support pinyon-juniper woodlands or grass.

Nutrient and Biomass Distribution

When assessing fire effects in chaparral, it is important to estimate biomass and nutrient distributions in plant and litter biomass because these materials serve as fuels and undergo various degrees of combustion during both prescribed burns and wildfires. Data on biomass and nutrients thus provide a basis for estimating changes in nutrient pools resulting from fire.

Data on plant biomass, litter and nutrient content is limited for Arizona chaparral. Some information is available on plant biomass, litter, and plant nutrients from a recent fire effects study being conducted on the Battle Flat watershed in the Prescott National Forest. Shrub cover on the 50-ha study watershed was determined using 16 30-m-long transects located randomly throughout the watershed. This watershed receives about 65 cm of precipitation annually and supports a dense shrub live oak-mixed shrub community. The area has not been burned for over 80 years (Dieterich and Hibbert, this volume). Weights of leaves, annual growth, small and large stems, and total plant biomass at Battle Flat were estimated using dimensional analyses of the major shrub species on 119 50x100 cm plots (table 1). The amount of biomass would be expected to vary from site to site, as has been reported for California

Table 1.—Biomass and nutrient distribution in a shrub live oak-mixed shrub community at Battle Flat on the Prescott National Forest. Data are means and standard deviations.

Ecosystem component	Biomass[1]	Nutrient			
		Nitrogen		Phosphorus	
	kg/ha	*%*	*kg/ha*	*%*	*kg/ha*
Leaves	4,060±314	1.82±0.40	74	[2]0.15	6
Annual growth	1,214±139	1.05±0.31	13	[2]0.12	1
Large and small stems	23,793±2090	0.53±0.19	126	[2]0.03	7
Litter	34,000	0.75±0.20	255	[2]0.10	34
Soil					
0-10 cm		0.11±0.04	1300	0.02±0.009	250
10-20 cm		0.09±0.03	1000	0.02±0.01	250

[1]*L. F. DeBano, USDA Forest Service, unpublished data.*

[2]*Estimated from P concentrations reported by DeBano and Conrad (1978) for California chaparral.*

chaparral (Riggan et al. 1988). However, the total biomass on this Arizona study site, 29,068 kg/ha, compared closely with the 30,394 kg/ha reported for a redshank *(Adenostema sparsifolium)* stand (Riggan et al. 1988), but was greater than the 19,000 kg/ha for scrub oak *(Quercus dumosa)* (Riggan et al. 1988) and 23,000 kg/ha for chamise *(Adenostema fasciculatum)* (Mooney et al. 1977) stands in Southern California. The average amount of litter under the shrub canopy at Battle Flat, about 34,000 kg/ha, was greater than the 16,400 kg/ha reported under oak-mountainmahogany chaparral on a central Arizona site receiving about 46 cm of precipitation annually (Pase 1972). Litter was estimated under shrub canopies on 382 100–cm^2 plots located randomly throughout the watershed. Compared with California chaparral, this amount of litter was greater than has been reported under chamise (20,500 kg/ha) and hairy ceanothus *(Ceanothus oliganthus)* (28,700 kg/ha) stands, but less than under scrub oak (54,300 kg/ha) and wartystem ceanothus *(Ceanothus crassifolius)* (45,500 kg/ha) stands (Riggan et al. 1988).

Information on plant and litter biomass at Battle Flat was used with nutrient concentration data to calculate total amounts of nutrients contained in different parts of the chaparral ecosystem (table 1). Nutrient estimates were limited to nitrogen (N) and phosphorus (P) because they are considered the two most limiting nutrients (Hellmers et al. 1955a, Vlamis and Gowans 1961). The biomass and nutrient profiles developed for Battle Flat must be used with caution for several reasons. First, data presented in table 1 represent only that area within a watershed covered by a shrub canopy and, as such, do not include the interspaces between individual shrub canopies. Therefore, when the data are used on a watershed basis, they would have to be reduced in direct proportion to percent area present as interspaces. Second, density of chaparral stands is known to vary widely and is highly dependent upon local precipitation regimes. Low-precipitation areas (those receiving less than 45 cm annually) will have much sparser stands of shrubs and, consequently, a given area would contain smaller amounts of nutrients than areas supporting dense brush. Third, shrubs are smaller on the more xeric sites and contain less plant biomass and litter than more mesic sites like Battle Flat.

The data in table 1 are, however, useful for illustrating several general characteristics of nutrient distributions in chaparral ecosystems that are important when assessing fire effects. First, a large proportion of the N and P pools is contained in the soil where it can be effectively insulated against heat generated during light-intensity fires. Second, surface litter is an important aboveground storage compartment for biomass and nutrients. Third, the leaves and annual growth contain higher concentrations of N and P than do the larger twigs, which is important, because most of these fine fuels are consumed during all intensities of fire.

Assessing and Predicting Fire Effects

Assessing the effect of prescribed burning, or wildfires, on soil nutrients requires estimating two things: (1) nutrients released, or lost, during combustion of standing plants (both live and dead) and surface litter layer; and (2) changes occurring in the upper mineral soil layers as a result of soil heating. Different techniques are required for assessing fire effects on each (combustion and soil heating effects). However, to understand either of these effects, it is important to be aware of the fire's unique behavior in chaparral.

Although fire behavior can be characterized in several ways, those indices related to rate of combustion and amount of aboveground biomass and litter consumed during a fire are probably best related to nutrient release from fuels and heating in the underlying soil.

Fire Behavior and Intensity

Large differences in fire behavior commonly observed between prescribed burns and wildfires in forests do not occur in chaparral. In forests, wildfires spread rapidly through the crowns of standing live and dead trees consuming much of the tree canopy (leaves, twigs, and, in some cases, boles) along with substantial amounts of surface needles and litter. Combustion of these woody fuels releases large amounts of thermal energy, which can produce substantial soil heating. On the other hand, prescribed fires in forests burn much less intensely because they are designed to consume only part of the surface needles and litter. These low-intensity fires are often referred to as "cool" fires. In contrast, both wild and prescribed fires in chaparral are carried through the shrub canopy. As a result, differences in fire intensity and soil heating during prescribed burns and wildfires in chaparral are not as great as those experienced in forests. For example, only minimal soil heating occurred during a cool-burning prescribed fire in a mixed conifer forest compared with substantial heating of the upper soil layers even during low-intensity chaparral fire (fig. 1A,B); soil surface temperatures never exceeded 100°C and the soil remained at ambient temperatures 5.0 cm below the soil surface (fig. 1A). In contrast, during a light-intensity fire in chaparral, the soil reached 250°C at the surface, and 100°C 2.5 cm downward in the soil (fig. 1B).

Fire intensities in chaparral can vary widely and, as a result, produce different degrees of soil heating (fig. 1B,C,D). For example, soil surface temperatures can reach 425°C during

moderately intense fires (fig. 1C) and about 700°C during intense fires (fig. 1D). Differences in surface temperatures are reflected in soil temperatures, which can reach 175°C in the 2.5-cm soil layer during a moderately intense fire and over 200°C during an intense fire.

Managers can reduce fire intensities when using prescribed fire in chaparral by specifying marginal burning conditions in their burn plan. If properly carried out, these less intense fires will consume only part of the shrub canopy and leave substantial amounts of unburned litter on the soil surface. However, although the entire canopy may not be consumed during a low-intensity fire, the remaining shrubs usually die and can contribute dead fuel for future fires. Recently developed aerial ignition techniques have also increased the chances of conducting prescribed burning during marginal, and safer, burning conditions. A better understanding of fire effects, combined with modern ignition techniques, allow managers to design burning prescriptions that can minimize fire intensities, and thereby reduce the impact on the soil resource.

Changes in Plant and Litter Nutrients During Combustion

Fire acts as a rapid mineralizing agent that releases plant nutrients from fuels (standing live and dead plants and surface litter) during combustion and deposits them in a highly available form on the soil surface (St. John and Rundel 1976, Vlamis and Gowans 1961). Substantial amounts of some nutrients, such as N, sulfur (S), and P, are lost into the atmosphere during combustion because the temperatures at which they volatilize are lower than the flaming temperature of woody fuels (1100°C) and, except for P, lower than glowing combustion temperature (650°C) (table 2). Nitrogen losses start at 200°C when organic matter is initially altered (Hosking 1938), S starts changing at 375°C (Tiedemann 1987), and P must reach 774°C before being lost (Raison et al. 1985). Loss of N is important because it is usually the most limiting plant nutrient in forest ecosystems (Maars et al. 1983), including California chaparral (Hellmers et al. 1955a, Vlamis and Gowans 1961). Phosphorus has also been reported to be limiting in forest ecosystems, particularly on areas where P-fixing soils are present (Vlamis et al. 1955). Results of controlled burning experiments show that the amount of total N volatilized during a fire is in direct proportion to the amount of organic matter combusted (Raison et al. 1985). In contrast, only about 60% of the total P is lost by non-particulate transfer when fuels are totally combusted. Percentage loss of S by volatilization is intermediate to that of N and P (Tiedemann 1987). Of the cations, potassium (K) is vaporized at the lowest temperature (774°C) (Raison et al. 1985); a measurable amount of K was reported lost during a prescribed fire in California chaparral (DeBano and Conrad 1978). Calcium, magnesium, and sodium are not vaporized until they reach temperatures of 1484°C (table 2) so most of these cations re-

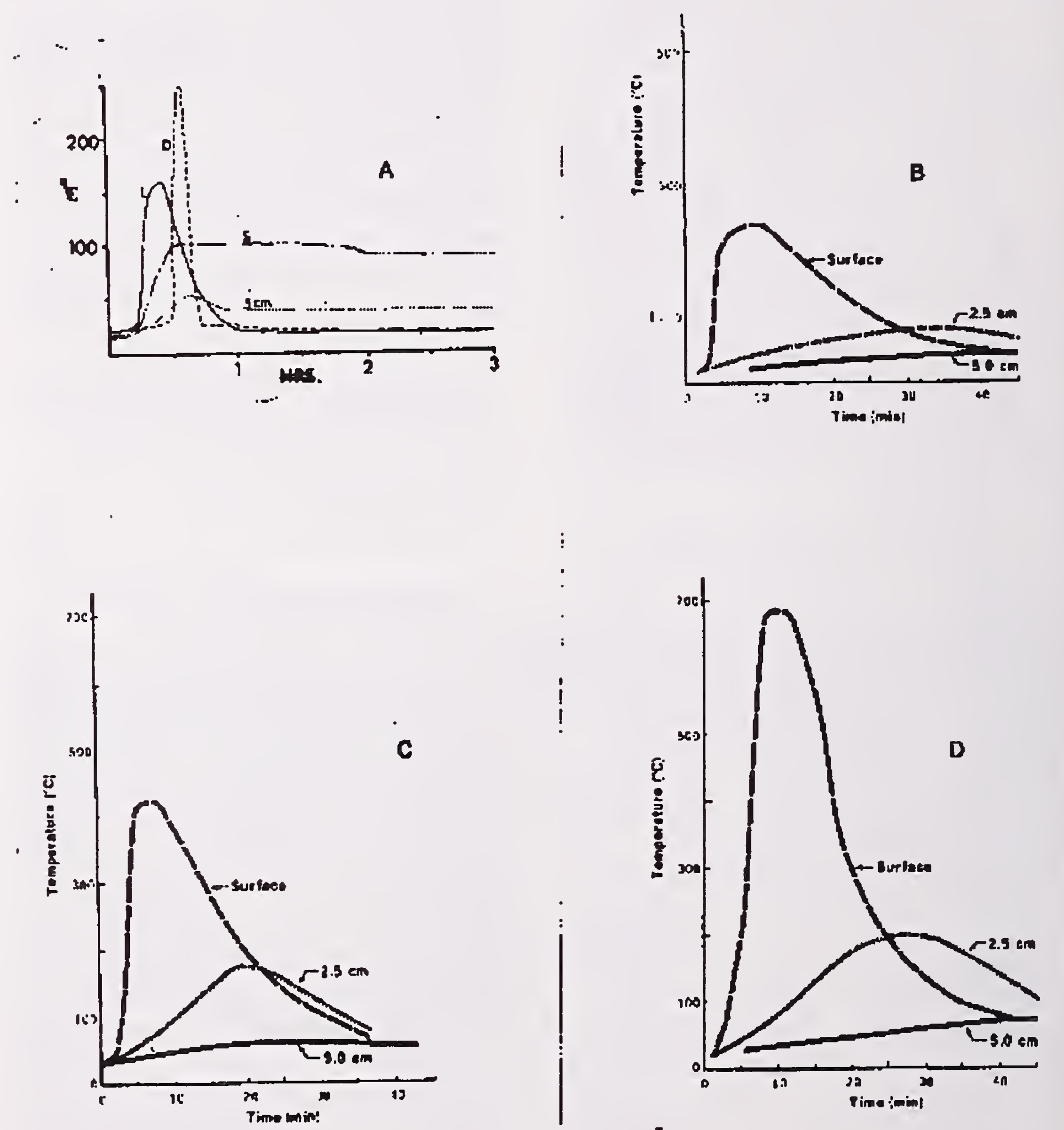

Figure 1.—Soil and litter temperatures during (A) a cool-burning prescribed fire in a mixed conifer forest (Agee 1973), (B) a light-intensity prescribed fire in chaparral, (C) a moderately intense fire in chaparral, and (D) an intense prescribed fire approaching wildfire conditions in chaparral (DeBano et al. 1979b). (L is litter temperature, D is duff temperature, surface is the temperature at the soil surface, 2.5 and 5.0 are soil temperatures 2.5 and 5.0 cm below the soil surface.)

leased during fire are deposited in the ash on the soil surface, except for small amounts that are transferred off-site in smoke (Clayton 1976).

A large body of information has been summarized on fire effects on soils (Wells et al. 1979) and fuels (Martin et al. 1979). Likewise, summary papers quantifying fire intensities are also available (Albini 1976, Rothermel and Deeming 1980). Unfortunately, because fire behavior is complex, a reliable mathematical model has not been developed that relates fire intensity measurements directly to soil heating, although some simplified models have been proposed for describing heat transfer in bare soils (Aston and Gill 1976) and for smothering litter (Albini 1975). Because adequate theoretical models are lacking, qualitative techniques have been used for characterizing fire intensity so its effect on soil properties can be evaluated. One such semi-quantitative method involves classifying fire intensity as light, moderate, or intense based on the visual appearance of burned brush and litter (Wells et al. 1979). This method is based on the assumption that soil surface conditions following burning are indicative of soil heating. A light burn occurs when the litter is singed and less than 40% of the brush canopy is burned. Irregular and spotty burning occurs and some leaves and small twigs remain on the plants either unburned or slightly singed. After a moderately intense burn surface litter is charred but not ashed. Between 40 and 80% of the plant canopy is burned by the fire and the remaining charred twigs are greater than 0.6-1.3 cm in diameter. After a severe fire only ashes remain on the soil surface and the area is completely burned. Plant stems remaining are 1.3 cm or greater in diameter. In many cases only charred remains of large plant stems are left on the burned area.

Table 2.—Threshold temperatures (°C (°F)) for insensitive, moderately sensitive, and sensitive soil properties.

Soil property		Threshold temperature[1]	
	Relatively insensitive		
Potassium and Phosphorus		774	(1425)
Calcium		1484	(3150)
Manganese		1962	(3564)
Clay destruction		980	(1796)
	Moderately sensitive		
Organic matter		100	(212)
Nitrogen		200	(392)
Sulfur		375	(707)
Soil structure		300	(572)
Soil wettability		250	(482)
	Sensitive		
Bacteria	—Wet	110	(230)
	—Dry	210	(410)
Nitrosomas bacteria	—Wet	75	(167)
	—Dry	140	(284)
Fungi	—Wet	100	(212)
	—Dry	155	(311)
VAM		94	(201)
Glowing combustion		650	(1202)
Flame temperatures		1100	(3542)

[1]*Temperature at which detectable changes of a soil property begin in response to*

Fire Effects on Mineral Soil

Developing guidelines for assessing fire effects on mineral soils can be visualized as a three-stage procedure, namely: (1) characterizing fire intensity; (2) relating fire intensity to fuel consumption, nutrient loss from fuels, and soil heating; and (3) finally predicting changes in chemical, physical, and biological soil properties resulting from nutrient input by ash deposition and soil heating of the mineral soil.

Once a burn has been classified as to its intensity class, soil heating can be estimated from curves developed by DeBano et al. (1979b) (fig. 1B, C, D). Temperatures in the soil can then be used for predicting changes from known responses (table 2) of soil properties to different soil heating regimes (DeBano 1979). Although these soil heating relationships were originally developed for California chaparral, they can serve equally well for Arizona because similar fuel loadings and fire behavior are found in both Arizona and California chaparral.

Fire effects on mineral soil can best be illustrated by using a conceptual model depicting an idealized soil profile that is exposed to surface heating by energy radiated downward during the combustion of plants and litter. Although most of the energy released during combustion is lost upward into the atmosphere, a small, but significant, quantity is absorbed by the soil surface and transmitted downward into the soil. It has been estimated only 8% of the total energy released during combustion of chaparral fuels is transmitted into the underlying soil (DeBano 1974). If heat radiates directly on a

bare, dry soil, it will be transmitted slowly into the soil with a minimum of soil heating because dry soil is a poor conductor of heat. Dry soil heats slowly because heat transfer is limited to particle-to-particle conduction and convection through soil pores. In contrast, wet soil conducts heat faster because it is also transferred rapidly by vaporization and condensation. There are also some differences in heat capacity of dry and wet soils, with wet soils absorbing greater quantities of heat per degree rise in temperature than dry soils because of the high specific heat capacity of water compared with mineral soil particles. Where thick litter layers are present, secondary combustion of litter occurs, further contributing to soil heating (Albini 1975). Conduction of radiated heat into the soil produces heating, which in turn affects the chemical, physical, and biological properties of the underlying soil.

Their spatial distribution in a typical soil profile expose some soil properties more directly to surface heating than others. For example, living organisms and soil organic matter concentrated on, or near, the soil surface are most affected. This strategic surface location exposes organic matter directly to heat radiated downward during burning. As a result, organic material and related soil properties are more likely changed by radiated energy than other soil properties, such as clay content, which is concentrated in subsurface layers where it is insulated from surface heating.

Soil chemical, physical, and microbiological properties strongly dependent on organic matter are susceptible to being changed by soil heating. For example, soil structure, cation exchange capacity, available nutrients, and microbial activity are all highly dependent upon organic matter, which begins changing chemically when heated to 200°C and is completely destroyed at 450°C (Hosking 1938). Cation exchange capacity of a soil depends not only on humus, but also on clay colloids. Humus is concentrated at, or near, the soil surface and thus is directly exposed to radiated heat. In contrast, clay formed by pedogenic processes is usually concentrated deeper in the soil profile, although clays can sometimes be found near the surface. Soil organic matter is also important for maintaining aggregate stability and soil structure, which in turn affects infiltration and other hydrologic properties of soils (e.g., water repellency, etc.). Soil chemical properties most readily affected are total and available forms of N, P, and S; and cation exchange capacity. Microbiological properties regulating input, loss, and availability of nutrients may also be significantly changed by soil heating.

Plant Nutrients

Although large amounts of total N and P are lost during the combustion of plants and litter, available ammonium-N and P are increased in the ash and upper soil layers during a fire (DeBano et al. 1979a). Ammonium-N is highest immediately after burning. Following burning, ammonium-N decreases and nitrate-N increases as nitrification occurs. The increased levels of available P produced during a fire slowly decrease and reach prefire levels within about one year. Highly available N and P provide a postfire fertility effect for newly established plant communities on burned sites. For example, a prescribed fire in Arizona increased ammonium-N concentrations in the surface 0-2 cm layer from 6 to 60 μg/g (which is an increase of about 15 kg/ha), did not affect nitrate-N concentrations of about 2 μg/g, and increased available P from 6 to 16 μg/g[3].

Soil Physical Properties

Soil physical properties dependent upon organic matter (e.g., soil structure and infiltration) are also affected directly by fire. An important hydrologic property affected by burning is soil wettability (DeBano 1981). During fires, organic matter in the litter and upper mineral soil layers is volatilized (fig. 2A). Most of the volatilized organic matter is lost upward in the smoke, but a small amount

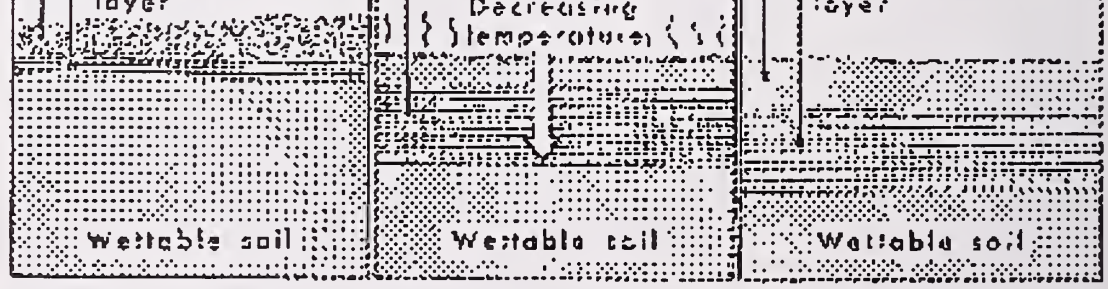

Figure 2.—Soil water repellency as altered by fire: (A) before fire, hydrophobic substances accumulate in the litter layer and mineral soil immediately beneath; (B) fire burns the vegetation and litter layer, causing hydrophobic substances to move downward along temperature gradients; (C) after fire, a water-repellent layer is present below and parallel to the soil surface on the burned area (DeBano 1969).

moves downward along steep temperature gradients in the upper 5 cm of the soil and condenses to form a water-repellent layer that impedes infiltration (fig. 2B, C) (DeBano et al. 1976).

The degree of water repellency formed depends on the steepness of temperature gradients near the soil surface, soil water content, and soil physical properties. For example, coarse-textured soils are more susceptible to heat-induced water repellency than fine-textured clay soils. The formation of this water-repellent layer, along with the loss of protective plant cover, increases surface runoff and erosion during the first rains following burning (DeBano and Conrad 1976). A reduction in infiltration by a water-repellent layer can lead to extensive rill erosion on burned watersheds (Wells 1981).

Soil Microbiology

Soil heating affects microorganisms either by killing them directly or altering their reproductive capabilities. Indirectly, soil heating alters organic matter, increasing nutrient availability and stimulating microbial growth. Although complex interrelationships exist between soil heating and microbial populations in soils, it appears that duration of heating, maximum temperatures, and soil water all affect microbial responses (Dunn et al. 1979, 1985). Microbial groups differ significantly in their sensitivity to temperature and can be ranked in decreasing sensitivity as fungi>nitrite oxidizers>heterotrophic bacteria (Dunn et al. 1985). Lethal temperature for bacteria was found to be 210°C in dry and 110°C in wet soil (table 2). Fungi tolerate temperatures of only 155°C in dry soil and 100°C in wet chaparral soils. Nitrifying bacteria appear to be particularly sensitive to soil heating, and even the most resistant *Nitrosomonas* bacteria can be killed in dry soil at 140°C and in wet soil at 75°C (Dunn et al. 1979). Physiologically active populations of microorganisms in moist soil are significantly more sensitive than dormant populations in dry soil (Dunn et al. 1985). Mineralization of killed microbial biomass may release readily available plant nutrients in addition to those produced directly by combustion of fuels.

Seed Mortality

Heat radiated during a fire also affects postfire germination of seeds occupying the litter and upper soil layers. Germination of seeds produced by some chaparral brush species is stimulated with exposure to elevated temperatures during a fire (Keeley 1987). Both maximum temperatures reached and time of exposure affect survival and germination of seeds. As with microorganisms, lethal temperatures for seeds are reached at lower temperatures in moist soils compared with dry.

Table 3.—Loss[1] of biomass, nitrogen, and phosphorus from vegetation, litter, and soil (in kg/ha) during light, moderate, and severe fires in Arizona chaparral.

Ecosystem component	Biomass Lgt	Biomass Mod	Biomass Int	Nitrogen Lgt	Nitrogen Mod	Nitrogen Int	Phosphorus Lgt	Phosphorus Mod	Phosphorus Int
Vegetation	11,628	17,440	26,160	121	151	188	5	8	9
Litter	5,100	17,000	30,600	38	128	230	3	10	18
Soil				0	65	130	0	0	2
Total				159	344	548	8	18	29

[1]The estimates of nutrient losses are based on data from under plant canopies (100% canopy cover) and therefore must be reduced in direct proportion to amount of interspace area present between shrubs.

Example of Assessing Fire Effects

The information given on nutrient losses during combustion of plants and litter along with estimates of soil heating can be used for assessing the impact of light, moderate, and intense fires on chaparral soils. The chaparral stand described for the Battle Flat watershed will be used for illustration.

According to the criteria cited, a light intensity fire leaves about 60% of the canopy unburned and only an irregular spotty burning of the surface litter occurs (Wells et al. 1979). Under these burning conditions it is reasonable to assume that only 10-15% of the litter will be lost. Assuming a 40% canopy reduction in canopy biomass, the shrub stand at Battle Flat would lose 11,628 kg/ha of canopy materials (table 3), of which 4,060 1,214, and 6,354 kg/ha would be lost from leaves, current growth, and large and small stems, respectively. This estimate is based on the complete combustion of leaves and current growth along with partial combustion of large and small stems Associated with the combustion of 11,628 kg/ha of canopy biomass would be a loss of about 121 kg/ha of N (assuming N loss is in direct proportion to biomass combustion), and 5 kg/ha of P (assuming only a 60% loss of P upon total combustion).

In addition to the N lost from the standing plants, an additional 15% of the litter N (38 kg/ha) and about 9% (0.60 X 15%) of the litter P (3 kg/ha) would be lost. Maximum soil temperatures would be about 275°C at the surface and 100°C at 2.5 cm in the soil (fig. 1B). Soil temperatures 5.0 cm deep in the soil would remain at

ambient temperature. No additional losses of N and P would be expected from the soil at these temperatures.

During a moderately intense burn most of the litter is charred, but not ashed. Between 40% and 80% of the plant canopy is consumed by the fire and the remaining charred twigs are greater than 0.6-1.3 cm in diameter (Wells et al. 1979). Under these conditions it is assumed the plant canopy is reduced 60% (17,440 kg/ha) and litter 50% (17,000 kg/ha) (table 3). Associated with a 60% combustion of canopy would be a loss of about 151 kg/ha of N and 8 kg/ha of P. An additional 128 kg/ha of N and 10 kg/ha of P would be lost during the combustion of 50% of the surface litter. Soil heating would reach about 425°C at the surface and be 175°C at 2.5 cm downward in the soil (fig. 1C). Moderately sensitive soil properties, including organic matter, N, S, soil structure, and soil wettability, would all be affected near the soil surface (table 2).

Substantial losses of N from soil organic matter could also occur. It is estimated 5% of the 1,300 kg/ha of N in the upper soil layer (65 kg/ha) could be lost (table 3). Soil P would not be expected to be lost because the surface maximum surface temperature (425°C) is below the threshold temperature for P (774°C) (table 2). However, at 2.5 cm only microbiological properties would be affected. Deeper at 5-cm in the soil, temperatures would only reach 75°C and have little, or no, effect on soil chemical, physical, and biological properties.

After a severe fire only ash remains on the soil surface and the only plant stems remaining are 1.3 cm or greater in diameter (Wells et al. 1979). In many cases only charred remains of large plant stems remain after the fire. Under these intense burning conditions it is assumed that at least 90% of both the plant canopy and surface litter are combusted. Large losses of N (188 kg/ha) and P (9 kg/ha) from the plants would occur along with greater losses from the litter (230 kg/ha of N and 18 kg/ha of P). Soil heating would be equally severe, reaching 700°C at the soil surface, 200°C and 2.5 cm, and 140°C at 5.0 cm. Near the soil surface severe damage would occur to all soil properties except K, P, calcium, manganese, and soil clays. Large amounts of soil N, perhaps 10% (130 kg/ha), and small amounts of P could be lost. The soil surface would most likely be sterilized, with no viable microorganisms or seeds surviving.

Using the above calculations it is estimated that the total N lost from plants and litter during light, moderate, and intense burns could be at least 159, 344, and 548 kg/ha, respectively. Additional losses of soil N may also occur with moderate and intense fires. Phosphorus losses from the plants and litter would be 8, 18 and 29 kg/ha during light, moderate, and intense fires, respectively. These calculations are in line with losses measured experimentally during chaparral fires in California where 149 kg/ha of N was lost during a prescribed fire when about 30% of the plants was consumed (a light-intensity fire). Amounts of N and P lost during burning will vary depending upon the amount of aboveground biomass, litter, and soil organic matter combusted during a fire.

It is important to remember that the above calcuations represent only changes under the plant canopy and therefore must be reduced in direct proportion to the amount of interspace area between shrubs when estimating total nutrient losses on an areal basis. However, the above calculations of N and P losses do illustrate the importance of burning chaparral as coolly as possible in order to reduce consumption of surface litter and soil heating. It is also important to be aware of these nutrient losses in order to insure that N-restoration to the site is included as part of postfire management.

Management Implications

The preceding discussion on fire behavior and fire effects on soils provides information that can be used when designing burning prescriptions or developing post-wildfire rehabilitation treatments.

Designing Burning Prescriptions

All available information on fire effects indicate it is desirable to burn as coolly as possible because this reduces litter combustion and soil heating. Although it is never possible to achieve the cool burning conditions prevailing during prescribed burning in forests, the intensity of chaparral fires can be mitigated to some extent. This requires burning under marginal burning conditions. Fire intensity in chaparral can be decreased to some extent by burning when (1) cooler air temperatures and higher humidities prevail, (2) live and dead fuel moistures are high, and (3) wind speeds are low. Fortunately modern ignition techniques make it easier to conduct successful prescribed burns under marginal burning conditions.

However, it must be kept in mind that even when marginal burning conditions prevail, localized sites within a prescribed burn area will burn intensely because "fire storms" are invariably generated by local wind convection patterns produced by the fire itself. Burning during marginal burning conditions is also more likely to produce a mosaic pattern of burned areas intermingled with unburned islands of vegetation. This mosaic pattern is less susceptible to postfire erosion than are large continuous areas. Also, reducing fire intensity reduces litter combustion, which, in turn, reduces total N and P losses from the site. Cooler burning conditions also reduce litter and soil surface temperatures, which lessen the chances of a water-repellent soil layer being formed. Reduced litter loss by combustion also protects the

soil surface from erosion by raindrop impact following burning.

Some studies suggest that burning over wet soils may not be beneficial to microorganisms and seeds because heating is more likely to be lethal in wet soils than in dry soils. However, it is not completely known what effect these changes in microbial populations have on nutrient cycling, decomposition, and site productivity. Nor is the effect that season of burning has on soils and nutrient cycling known. However, it is known that nutrient content of plants change throughout the year. Higher concentrations of N and P are present in the leaves and small fuels during active growing seasons. Burning during periods of active growth would potentially expose larger quantities of N and P to combustion losses.

Although managers are not always able to burn under marginal burning conditions, they still need to be able to assess changes in nutrient relationships, erosion, and site productivity resulting from prescribed fires. Sufficient research information is currently available for making reasonable estimates of the changes that can be expected to result from different burning prescriptions.

Postfire Rehabilitation

Postfire rehabilitation plans must consider both short- and long-term fire effects. Some important considerations are losses of total nutrients (particularly N), changes in nutrient availability, decreased infiltration rates, and subsequent erosion on burned areas.

Nutrient Loss and Replenishment

Although several plant nutrients are lost directly by combustion during fire and by erosion following fire, N is the most important nutrient because it is lost in the largest quantities and is likely to be the most limiting nutrient in chaparral ecosystems (Hellmers et al. 1955a). Postfire rehabilitation plans must consider natural mechanisms available for replenishing N in order to assure long-term productivity of these chaparral ecosystems.

Several mechanisms are available for restoring N lost during a fire. These include input by bulk precipitation and N-fixing plants and microorganisms. Bulk precipitation is estimated to restore about 1.5 kg/ha annually, which is not sufficient to restore the N lost if it is assumed chaparral burns every 25-35 years (Ellis et al. 1983). However, the annual input of N may be substantially greater in localized areas having large amounts of airborne N pollutants present (e.g., Los Angeles Basin). Riggan and coworkers (1985) found annual inputs of 23.3 and 8.2 kg/ha of N as canopy throughfall and bulk precipitation, respectively. It is unlikely that significant atmospheric input of N occurs in Arizona chaparral ecosystems because most of the areas supporting this shrubby vegetation are far removed from metropolitan centers.

An important source of N replenishment appears to be by N-fixing organisms. Nitrogen fixation by short-lived legumes is important in postfire management in California chaparral, but is absent in Arizona. Nitrogen fixation by asymbiotic organisms is also low, amounting to about 1 kg/ha annually. It now appears the most likely source of ecosystem N is biological N-fixation by actinomycete nodulated shrubs such as *Cercocarpus betuloides*. Therefore, it becomes important in postfire planning to manage for the reestablishment of these N-fixing shrubs, which can effectively fix N after the "flush" of available N produced by the fire has been immobilized. Both ammonium- and nitrate-N generally drop to prefire levels within a year after fire.

Another postfire rehabilitation treatment that affects N-fixation is competition between introduced plants used for erosion control and the reestablishment of native plants. Although dense stands of seeded grasses can become established following fire, it is the exception, rather than the rule, because postfire grass reseeding is unpredictable. However, when dense stands of grass become established they can effectively compete with shrubs for moisture and nutrients. For example, it was found in California that reseeded annual ryegrasses competed with seedling establishment of N-fixing shrubs (e.g., *Cercocarpus* sp. and *Ceanothus* sp.), and even with sprouting species (Conrad and DeBano 1974). Therefore, it is recommended that management plans concerned with rehabilitating burned areas address the establishment and perpetuation of N-fixing shrubs during the postfire plant reestablishment period.

Postfire Fertilization

The question frequently arises whether there is a need to fertilize as part of postfire rehabilitation. Fertility trials have generally shown that burned soils have a greater supply of available N than unburned soils (Vlamis et al. 1955). Likewise, N fertilizer responses were not detectable on field plots immediately following a chaparral fire in southern California (DeBano and Conrad 1974). Postfire responses to P fertilizers are more variable because some soils can rapidly fix available P produced during burning (DeBano and Klopatek 1988, Vlamis et. al 1955). The preponderance of research findings indicate fertilization is probably not a desirable treatment immediately following burning. In fact, fertilization may have a depressing effect on N-fixation because additional amounts of highly available N are added to already high levels produced by burning. Also, the high levels of available N following fire could increase denitrification. The advisability of P fer-

tilization is less clear but it may be of little advantage in those soils that fix available P. In summary, it appears that fertilizing in the "ash" is not a recommended postfire treatment for at least one year following burning.

Postfire Erosion

There are limited opportunities for preventing, or reducing, erosion on chaparral soils burned under wildfire conditions. Grass reseeding has been widely used as a postfire rehabilitation treatment. The usefulness of grass reseeding for reducing postfire erosion has not been clearly established because grasses have only a limited opportunity to become established before active erosion occurs during the first rainy season following fire. It is also extremely difficult to design studies clarifying the relationship between grass establishment and erosion because of the high variation encountered under field conditions (Barro and Conard 1987). Grass competition may also indirectly interfere with the establishment of native plants following fire and, as a result, increase long-term erosion. Establishment of a dense grass cover on burned sites may also increase the volume of fine dead fuels by the end of the first growing season, thereby making these areas more susceptible to ignition and early reburn.

The judicious use of prescribed fire provides a potential technique for minimizing large erosional losses following wildfires. In southern California prescribed fire is being advocated as a tool for reducing wildfire severity by creating uneven-age stands that break up continuous fuel loads necessary for sustaining large-scale wildfires (Florence 1987). Reducing the probability of intense, widespread wildfires by cooler burning prescribed fires would be expected to reduce fire impacts on soil properties and postfire erosion. This management concept is also compatible with the concept of developing brush-grass mosaics for water augmentation in Arizona chaparral (Bolander 1982).

Concluding Comments

Both wild and prescribed fires occur frequently in Arizona chaparral. These fires affect physical, chemical, and biological soil properties of soils. Prescribed fires in chaparral are generally more intense than those in forests because fire is carried primarily through the shrub canopy. Therefore, more subtle differences in fire behavior exist between wildfires and prescribed burns in chaparral, although burning prescriptions in chaparral can be designed to reduce total fuel consumption, thereby minimizing soil heating and associated fire effects.

Soil chemical, physical, and microbiological properties most strongly interrelated with organic matter are most susceptible to being changed by soil heating. For example, soil structure, cation exchange capacity, available nutrients, and microbial activity are all highly dependent upon organic matter, which begins changing chemically when heated to 200°C and is completely destroyed at 450°C. Fire also acts as a rapid mineralizing agent; it releases plant nutrients from fuels which are then deposited on the soil surface in a highly available form. During combustion, however, substantial amounts of N, S, and P can be lost to the atmosphere. Replenishment of N losses should be included in postfire rehabilitation plans. Treatments interfering with the establishment of postfire N-fixing plants should be avoided. A particularly important concern is the possible competition between reseeded grasses and native N-fixing plants.

Burning increases the availability of most plant nutrients. Although total N is lost, available ammonium-N and P increase substantially during burning. The presence of high concentrations of available plant nutrients on the soil surface immediately following fire negates the advantage of fertilizing for at least one year after burning.

In the final analysis, the judicious use of prescribed fire has an important role in managing chaparral ecosystems in Arizona. Prescribed fire can be used as a technique for reducing potentially catastrophic wildfires. Improved wildlife habitat, better access, and increased water production also result from well-planned prescribed burning programs. However, certain precautions must be taken when planning postfire treatments to assure the continued long-term productivity of this unique ecosystem.

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Effects of Fire on Pinyon-Juniper Soils[1]

W. Wallace Covington[2] and Leonard F. DeBano[3]

Abstract—The purpose of this paper is to synthesize published information on fire effects in pinyon-juniper and other forest and woodland types to draw inferences about likely effects on soils in pinyon-juniper woodlands in the Southwest. In contrast to many other types, fire effects in pinyon-juniper vary greatly from point to point because of the spatial heterogeneity of the fuels.

Pinyon-juniper woodlands occupy between 25 and 32 million ha in the western United States (Arnold et al. 1964, Clapp 1936, Kuchler 1964,) and more than 5.7 million ha in Arizona and New Mexico (Springfield, 1976). Pinyon-juniper woodlands in Arizona and New Mexico are located at elevations from 1300 to 2300 m. These woodlands are characterized by a mosaic of pinyon and juniper trees surrounded by interspaces, occupied by sparse to dense herbaceous and shrubby vegetation. The principal tree species are: pinyon pine (*Pinus edulis*), Mexican pinyon (*P. cembroides* Zucc.), Utah juniper (*Juniperus osteosperma*), one-seed juniper (*J. monosperma*), alligator juniper (*J. deppeana*) and Rocky Mountain juniper (*J. scopulorum*). The dominant understory species in southwestern pinyon-juniper is blue grama [*Bouteloua gracilis* (H.B. K.) Lag.] although other herbaceous and shrubby plants may be present (Pieper 1977, Springfield 1976). These pinyon-juniper woodlands occur in the transition zone between semiarid vegetation (i.e., chaparral, desert shrub or grasslands) and coniferous forests.

Not only do pinyon-juniper stands vary widely in structure and species composition, but also they occupy soils derived from a broad range of parent materials including granite, basalt, cinders, limestone, sandstone, and mixed alluvium (Aldon and Brown 1971). Soils vary in texture from stony, cobbly and gravelly sandy loams to clay loams and clay, and vary in depth from shallow to deep (Springfield 1976, Pieper 1977).

In pinyon-juniper woodlands, a soil nutrient mosaic pattern develops where carbon, nitrogen (N), and available phosphorus (P), are concentrated in the upper soil layers under the tree canopy. This pattern reflects the accumulation of litter by different plant species (Barth 1980, Charley and West 1975, Everett et al. 1986, Lyons and Gifford 1980, Klopatek 1987). Tree growth rates vary widely between sites in close proximity to one another. Although N is usually considered the most limiting nutrient in forest ecosystems (Maars et al. 1983), it appears P and potassium (K) may also be limiting in pinyon-juniper ecosystems (Barrow 1980, Bunderson et al. 1985).

Wildfires were common in the pinyon-juniper type before European settlement (Leopold 1924) and probably restricted the establishment of pinyon-juniper woodlands in savannas to isolated stands of trees occupying shallow, rocky soils which would not support grasses (Burkhardt and Tisdale 1969, O'Rourke and Odgen 1969, Johnsen 1962). Heavy grazing in the late 1800's and early 1900's eliminated fuel continuity, and this, combined with an active fire suppression policy, decreased fire occurrence throughout this type (Wright et al. 1979). Fire exclusion, coupled with decreased grass competition, may have encouraged tree invasion into former grasslands (Wright et al. 1979).

Prescribed burning has been used in pinyon-juniper in four situations: (1) broadcast burning, (2) burning individual trees, (3) burning grassland areas, and (4) burning slash (Arnold et al. 1964). The first three burning situations are used mainly for range improvement. Slash burning is used both during range improvement and after fuelwood harvesting. However, fire severity (used here to mean relative heating and impact on ecologic processes) and behavior and its effect on the soil resource is very different under these four burning situations. Very severe fires are prescribed to consume as much of the tree as possible if the objective is to kill mature trees. Grassland areas are burned primarily to kill invading seedlings and younger trees; for this, the fire is mainly a light fire that is carried by the grass cover. Slash disposal during range improvement programs involves burning mature trees that have been killed mechanically or chemically. If the slash is piled and burned, severe soil heating can occur because large

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[2]*Professor, School of Forestry, Northern Arizona University, Flagstaff, AZ 86011.*

[3]*Principal Soil Scientist, Rocky Mountain Forest and Range Experiment Station, Stationed at the Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287*

amounts of fuel are burned on small areas. Some sites east of Flagstaff, where slash has been pushed into piles with bulldozers and then burned have remained devoid of vegetation for well over twenty years (personal observation). When the slash is broadcast and burned, fuel loading is less and fire effects are spread out over a larger area. If these fine fuels (small diameter, i.e., <1-2 inches in diameter, branches, twigs, and needles) are piled, intense fires may severely affect small localized areas of soil under the piles.

In summary, fire effects in pinyon-juniper are not well understood. Pinyon-juniper woodlands vary widely both geographically and from point to point within a particular location. Although some research has been reported on nutrient relationships in pinyon-juniper soils (Barth 1980, Bunderson et al. 1985), few studies have been done on the effects of burning on nutrient availability or other soil characteristics (DeBano et al. 1987, Gifford 1981). The purpose of this paper is to synthesize published information regarding the effects of fire, both wildfire and controlled burning, on soils in pinyon-juniper woodlands in the Southwest. The intended audience for this paper is managers and others interested in fire management of pinyon-juniper woodlands. For fire effects on some soil properties, there are no studies in pinyon-juniper woodlands. In these cases we have drawn inferences from studies in other types and from our understanding of general principles relating to fire effects in forests and woodlands. However, the reader should be cautious about an overreliance on research based on other types.

Related Literature

Several review papers serve as useful references for fire effects in pinyon-juniper woodlands. Wells et al. (1979), in the state-of-knowledge review of fire effects on soil, present an excellent overview of fire effects on soil in general and contains several references to pinyon-juniper woodlands. In a parallel publication, Lotan et al. (1981) review the effects of fire on flora, including two pages on pinyon-juniper woodlands. Fire effects on fuels have been summarized by Martin et al. (1979), in that same state-of-knowledge series. The final review paper we recommend that managers read is by Wright et al. (1979), in which they review the role and uses of fire in pinyon-juniper woodlands as well as sagebrush-grass communities.

Fire Effects In Pinyon-juniper

Nutrient Distribution in Pinyon-Juniper Woodlands

Pinyon pine and juniper trees cycle nutrients both horizontally (Tiedemann 1987) and vertically (DeBano et al. 1987). Tree roots penetrate into interspace soils between tree canopies where they absorb nutrients and incorporate them into tree biomass. A large portion of the nutrients captured from interspaces are deposited on the soil surface under trees during leaf fall, where they are released in an available form by decomposition, thereby enriching the upper soil layers. Trees also translocate nutrients vertically to the soil surface from deeper in the soil profile by a similar process. However, little is known about variations in the quantity of nutrients cycled by trees.

Published information on nutrient patterns in pinyon-juniper woodlands clearly portrays strong vertical and horizonal distribution patterns developing from the above described nutrient cycling processes. The most important vertical compartments are: aboveground biomass, litter, and soil. Nutrients are also distributed and exchanged horizontally between trees and interspaces. The quantities of nutrients stored in soils under tree canopies have been reported by some authors to be greater than those in interspaces (Everett et al. 1986, Tiedemann 1987), but in other cases no significant differences could be detected (DeBano et al. 1987). We used information presented in the literature on N, P, and K to develop a model portraying vertical nutrient distribution patterns under trees and in associated interspaces for a typical pinyon-juniper ecosystem (table 1). Data on N presented in table 2 for a pinyon-juniper ecosystem were taken from Tiedemann (1987). Distributions of P and K were taken from DeBano et al. (1987) where soils data for the 0-3.8-cm depth was extrapolated linearly to 60 cm.

Important features of the vertical distribution pattern are: (1) less than 20 percent of the total nutrient pool resides in plant biomass and litter, and (2) differences in the proportion of a nutrient stored in living biomass and litter were present among the three nutrients. For example, under tree canopies higher proportions of N are present in litter and aboveground biomass compared to P and K. About 98 percent of total P in the tree dominated patches is contained in soil compared to 93 percent for K, and 82 percent for N. Horizontally N, P, and K are concentrated in a mosaic pattern corresponding to litter and canopy distribution. Although total N, P, and K in the soil may, or may not, differ significantly beneath tree canopies and interspaces, more total N, P, and K accumulates in live tree boles, stems, and leaves and in litter under tree canopies than in interspace vegetation (DeBano et al. 1987).

Nutrient Availability

A delicate balance exists between available and total nutrients in unburned pinyon-juniper woodlands because only a small percent of the total nutrient pool is in a readily

available form. Available nutrients exhibit vertical and horizontal distribution patterns similar to total nutrients. Horizontal patterns of nitrate and ammonium are influenced by tree canopy distribution, with higher concentrations of ammonium being found in the surface soil under tree canopies compared to interspaces (DeBano et al. 1987). In contrast, nitrate may or may not differ between trees and interspace areas (DeBano et al. 1987, Klopatek 1987, Thran and Everett 1987).

Amount and Consumption of Fuels

Little data are available for fuels and fuel consumption during prescribed burns although some qualitative estimates have been made of fuel consumption during wildfires (Martin et al. 1979). A study of sixteen wildfires in Arizona showed that fires occurring on flat to gently rolling terrain tend to burn intensely — consuming all available ground fuels, killing most of the trees, and leaving the dead skeletons of the trees standing (Arnold et al. 1964). Natural ground fuels are rarely heavy in pinyon-juniper stands and probably never exceed 1 to 3 tons per acre. Wildfires occurring during strong winds, low relative humidities, and air temperatures above 90° F can destroy nearly all the trees and remove all the understory vegetation and litter (Arnold et al. 1964).

Nomograms have been developed for singleleaf pinyon (*Pinus monophylla*) and Utah juniper (*Juniperus osteosperma*) trees which relate different-sized fuels quantitatively to crown area (Meeuwig et al. 1979). These relationships were originally developed for estimating biomass and fuel loading from aerial photos. Although these nomograms were developed in Utah and Nevada, Utah juniper is an important component of southwestern pinyon-juniper woodlands making these relationships useful for estimating fuel consumption from wildfires or prescribed burning used as part of fuelwood harvesting and range improvement projects. However, it must be recognized that these fuel values are only for fuels directly under the canopies of the trees. According to these nomograms, the distribution for a typical pinyon-junper stand having trees with a canopy diameter of 6.5 meters (33 square meters) would be as given in table 2.

The fuel distribution presented in table 2 indicates that during severe wildfires where all the tree and litter are consumed by fire, about 19 kg/m^2 of pinyon and 11 kg/m^2 of juniper biomass would be lost, in addition to 7.5 kg/m^2 of litter. Intense prescribed fires burning under extreme conditions would be similar to wildfires and could also consume nearly all the plant and litter biomass. Likewise, the same level of biomass loss

Table 1. Amounts of nitrogen, phosphorus, and potassium (kg/ha) in aboveground biomass and the 60-cm soil depth under trees and associated interspaces in pinyon-juniper woodlands and percent in each ecosystem compartment.

Ecosystem compartment	N[1]	P[2]	K[2]
	Trees		
Foliage	108 (1)[3]	20 (<1)	147 (4)
Twigs	184 (2)	11 (<1)	55 (1)
Wood	133 (2)	5 (<1)	21 (<1)
Litter	1000 (12)	44 (1)	65 (2)
Soil (0 - 60 cm)	6615 (82)	3963 (98)	3584 (93)
Total	8040	4043	3872
	Interspaces		
Foliage	4 (<1)	.2 (<1)	.2 (<1)
Soil	4527 (>99)	3963 (>99)	3584 (>99)

[1]Data from Tiedemann 1987
[2]Data from DeBano et al. 1987
[3]Percent of total nutrient pool made up by a particular ecosystem compartment.

Table 2.—Amount (kg/m^2) of different-sized fuels and litter directly under the canopies in pinyon and juniper trees having an average canopy diameter of 6.5 m (from Meeuwig et al. 1979).

	Pinyon		Juniper	
Fuel	Amount[1]	Percent[2]	Amount[1]	Percent[2]
Foliage	2.0	10	2.5	23
Stems <.64 cm	2.0	10	1.0	9
Stems (.64-2.5 cm)	2.0	10	1.0	9
Stems (2.5-7.6 cm)	3.0	20	2.0	23
Stems (>7.6 cm)	9.5	50	4.0	36
Litter	7.5		7.5	

[1] Number represents kg/m^2 of tree canopy area.
[2] Number is percent of the total tree biomass.

would occur during a fuelwood harvesting operation if all the slash were completely burned and surface litter destroyed. There are opportunities to reduce nutrient losses during a fuelwood operation by burning on interspaces instead of over litter accumulations produced by former tree canopies. It is also possible during slash burning operations to burn under cooler burning conditions so that not all the slash is consumed. Under the most conservative slash burning operation most of the foliage and stems less than 2.5 cm in diameter would most likely be consumed, leaving about 20 percent of the total plant biomass on the site.

Effect of Heating on Soil Properties

The spatial distribution of soil properties in soils under pinyon-juniper woodlands makes some soil properties more vulnerable to surface heating than others. Living organisms and soil organic matter are concentrated on, or near, the soil surface and decrease exponentially with depth. This surface location exposes organic matter directly to heat radiated downward during burning. As a result, organic material and related soil properties are more likely changed than those concentrated in subsurface layers which are insulated against surface heating.

Soil chemical, physical, and microbiological properties strongly dependent on organic matter are susceptible to being changed by soil heating. For example, soil structure, cation exchange capacity, available nutrients, and microbial activity are all highly dependent upon organic matter, which begins changing chemically when heated to 200°C and is completely destroyed at 450°C (Hosking 1938). Soil organic matter is also important for maintaining aggregate stability and soil structure, which, in turn, affects infiltration and other hydrologic properties of soils. Microbiological properties most affected by heating are: heterotrophic bacteria, nitrifying bacteria, fungi, and mycorrhizae.

Effects on Nutrients Contained in Plants and Litter

The immediate effects of fire on nutrient content of the soil are the result of the interaction of nutrient volatilization and mineralization of nutrients stored in organic matter. Nutrients are volatilized at different temperatures, with N, P, and S having low volatilization temperatures and Ca, Mg, and K having relatively high volatilization temperatures (see DeBano, this volume). Thus, burning volatilizes much of the N, P, and S, but Ca, Mg, and K are left behind in ash, primarily as oxides.

Information on nutrient distributions from tables 1 and 2 can be combined with estimates of fuel consumption and used to assess the effect of fuelwood harvesting, or range improvement activities, on nutrient cycling and loss. Fuelwood harvesting alone, without using prescribed fire, would remove about 133 kg/ha of N in the woody material. However, if prescribed fire was used for slash disposal following harvesting, an additional 277 kg/ha of N would be volatilized from twigs and leaves (assuming 95 percent of the N is volatilized) in addition to variable amounts of the 1000 kg/ha of N contained in the litter. If large amounts of litter were consumed by fire during slash disposal operations then an additional loss of up to 400-500 kg/ha of N could occur. Fuelwood harvesting would have a lesser impact on P and K because it would remove only a small percentage of the P (5 kg/ha) and K (21 kg/ha). Substantial losses of P would also occur if the leaves and twigs were burned following fuelwood harvesting. Nonparticulate losses up to 50% of the total P (16 kg/ha) could occur if these fine materials were totally consumed during burning (Raison et al. 1985). A variable amount of the P contained in the litter could also be lost depending on the intensity of the fire. Similar percentages of K may also occur because it volatilizes at the same temperature as P (Raison et al. 1985).

Effects on Nutrients in Soil

In addition to the immediate effects of fire on soil nutrients, fire may change the rates of: (1) nutrient mobilization (because of changes in allelopathic chemicals, moisture balance, temperature, and general soil chemistry), (2) nutrient leaching (through changes in vegetation cover and infiltration rates), and (3) nitrification (because of high ammonium substrate available for conversion to nitrate).

Since N and P have been shown to be the most limiting nutrients for a wide variety of forest and woodland types, we will concentrate our review on those elements.

Soil Chemical Properties and Nutrient Availability

Although large amounts of total nitrogen and phosphorus are lost during the combustion of plants and litter, available forms of these nutrients are higher in the ash and upper soil layers following a fire (Christensen 1973, DeBano et al. 1979, DeBano et al. 1987, DeBano and Klopatek 1988).

In southern Utah, slash burning in pinyon-juniper has been shown to increase soil P, total Kjeldahl N (TKN), K, and percent organic matter under the burned slash piles to a depth of 4 inches (Gifford 1981). Covington et al. (1986) found that slash burning caused immediate increases in ammonium proportional to the amount of fuel consumed. DeBano et al. (1987) found that slash burning in central Arizona caused a shift from ammonium to nitrate.

Covington et al. (in review), using a time series approach, found that most of the changes in N were in the first 5 years after slash burning in northern Arizona pinyon-juniper. During the first year ammonium increased from 1 ppm on unburned sites to over 100 ppm under burned slash piles. Nitrate was little changed at first; however, by year 2 after burning nitrate was over 50 ppm on burned sites and less than 0.5 ppm on unburned sites. These changes over time were attributed to a lag in the nitrification process. By 5-7 years after burning, ammonium and nitrate had returned to pretreatment levels.

Inorganic P is also released by burning, but it also is quickly immobilized chemically (DeBano and Klopatek, 1988) and may no longer be readily available for plants. The increased levels of extractable phosphorus produced during a fire slowly decrease and reach prefire levels in about one year. Highly available nitrogen and phosphorus provide a post-fire fertilizing effect until plant communities become established and are able to utilize this supply of available nutrients.

Soil pH Changes

Soil pH in some forest types has been shown to increase after burning (e.g., Grier 1975, DeByle and Packer 1976) because of the depostion of Ca, Mg, and K oxides and their subsequent hydration into bicarbonates. As these bicarbonates are leached into the mineral soil they can cause an increase in pH up to 7 or so. Heating has also been shown to increase pH in some soils (Tarrant 1956). However, in soils which approach a neutral pH, such as occur throughout most of the pinyon-juniper type, no meaningful change in pH would be expected. This has been shown to be the case in southwestern ponderosa pine following prescribed burning, where both burned and control sites had pH's of 6.2-6.5 (Ryan and Covington 1986), near the equilibrium pH for bicarbonate.

Soil Moisture Relations

Burning can cause changes in soil moisture relations through its effects on interception by vegetation and litter, evaporation, transpiration, infiltration, and soil moisture tension. Since we could find no studies of fire effects on soil moisture relations in pinyon-juniper, we will have to rely on results from other types. However, again we caution the reader that these inferences should be regarded as tenuous until research in the pinyon-juniper type can be done.

Changes in interception of precipitation and transpiration would be greatest when burning kills pinyon and juniper trees and they are replaced by herbaceous vegetation, which has a lower plant surface area. The removal of tree cover, and the forest floor material occurring under the tree canopy, substantially reduces the interception surface of the ecosystem (Anderson 1976, Campbell et al. 1977). Reductions in interception would reduce evaporation and, all other things being equal, increase the amount of precipitation delivered to the soil surface.

Since interception plays such an important role in controlling evaporation from most arid and semiarid forest and woodland types, the net result should be an overall decrease in evaporation. However, higher soil temperatures resulting from the change in albedo (darker soil surface) most likely increases evaporation from the soil surface.

Once the water is delivered to the soil surface, the next factor is whether it infiltrates into the soil body or runs off as overland flow. Buckhouse and Gifford (1976) found decreased infiltration on pinyon-juniper sites where chaining debris had been burned and the sites had been seeded and grazed. However, research from other forest types shows a wide range of responses to burning, from decreases in infiltration (e.g., Zwolinski 1971, McMurphy and Anderson 1965), to no change (e.g., Veihmeyer and Johnson 1944), to increases (e.g., Scotter 1964). The extent to which water repellency may be involved in decreasing infiltration losses is unknown. However, based on work in chaparral (DeBano et al. 1976) and in southwestern ponderosa pine (Campbell et al. 1977), one would expect water repellency on coarse textured soils (with less than 5% clay [DeBano et al. 1970]) under slash which has been burned. Although it has not been reported in burned stands, Scholl (1971) did observe a water repellent layer immediately beneath the litter of Utah juniper (*Juniperus ostesperma*); thus the potential for fire-induced water repellency in pinyon juniper is likely. Nonetheless, since reports are so variable, it would be best to withhold judgement regarding the likely effects of burning on infiltration in southwestern pinyon-juniper woodlands for the time being.

The next factor is whether fire alters the capacity of the soil to retain water. Water holding capacity may be reduced by burning in some forest types, depending on the fire intensity. Where heavy fuels burn (e.g., slash piles) one would expect a reduction in water holding capacity in pinyon-juniper. However, the net effect on soil moisture content varies widely (Wells et al. 1979). Soil moisture tension might be expected to increase after burning in proportion to the amount of fuel consumed because of the increased ions leached from the ash. Finally, fire may reduce the amount of water taken up by plants by reducing the transpiring surface of the vegetation through mortality.

Soil Temperature

Soil temperatures during burning vary with depth and are affected by

fuel load, fuel moisture content, soil moisture content, and soil texture. The greatest heating and the highest temperatures would occur where heavy, dry fuels (e.g., slash piles) are burned over dry, coarse textured soils. However, the temperature gradient is typically steep.

For example, in relatively cool prescribed burns in ponderosa pine and incense cedar, duff temperatures were approximately 260 degrees C, surface temperatures reached only 93 degrees C, and at 2 inches the temperature was barely affected (Agee 1973). By comparison burning heavy windrowed eucalyptus slash and logs caused temperatures of 666 degrees C just below the soil surface and 112 degrees C at a depth of 8.5 inches (Cromer 1967, Cromer and Vines 1966).

Pinyon-juniper slash burning should be intermediate between these two examples in its effects on soil temperatures.

Effects on Soil Microbes

Soil heating directly affects microorganisms either by killing them directly or altering their reproductive capabilities. Indirectly, soil heating alters organic matter, which increases nutrient availability, and stimulates microbial growth. Although complex interrelationships exist between soil heating and microbial populations in soils, it appears that duration of heating, maximum temperatures, and soil water all affect microbial responses (Dunn et al. 1979, 1985).

Microbial groups differ significantly in their sensitivity to temperature and can be ranked in decreasing sensitivity as: fungi>nitrite oxidizers>heterotrophic bacteria (Dunn et al. 1985). Fire has been reported to decrease vesicular-arbuscular mycorrhizae propagules in pinyon-juniper soils when soil temperatures reached 60°C or greater (Klopatek et al. 1988).

Summary and Management Implications

Before European settlement, wildfires burning at 10- to 30-year intervals restricted pinyon and juniper trees to rocky soils and outcrops throughout much of what today is extensive pinyon-juniper woodlands. Heavy grazing and active fire suppression allowed the encroachment of pinyon and juniper seedlings into what had previously been grassland.

Pinyon-juniper woodlands vary substantially widely from location to location and from point to point within a location. Fire is being used in pinyon-juniper woodlands for a variety of purposes including: (1) eradication of pinyon-juniper from areas of grassland it has invaded, (2) broadcast burning of fuelwood harvesting slash, and (3) burning of piled slash. Thus fire effects vary widely, depending upon the characteristics of the pinyon-juniper area being burned and the type of burn.

Generally speaking, the greatest impacts occur where fuel loads and fuel consumption are the greatest. Fuel loads in uncut stands are heaviest directly under the tree canopies. In harvested stands, fuel loads are greatest where slash is piled on top of canopy litter. Fuel loads are somewhat less when slash is piled in the interspaces, and much less when slash is broadcast. Fuel loads are least where herbaceous vegetation is dominant. Thus, the greatest impacts of fire in pinyon-juniper are very localized, occurring immediately under burned slash piles and burned canopy litter.

As is the case for other forest and woodland types, fire effects are the greatest at the soil surface, declining exponentially with depth. Burning volatilizes varying amounts of N and S (Tiedemann 1987), which are critical nutrients in limiting ecosystem productivity. However, cations (Ca, Mg, and K) as well as some P, N, and S are left behind in the ash. The nutrients in this ash plus some N as ammonium may be transferred into the soil thereby increasing soil fertility (the ashbed effect).

Although we could find no research on fire effects on soil moisture or temperature, some inferences can be drawn from research in other forest types. Soil moisture relationships after burning may be improved, degraded, or have no effect.

Soil temperatures produced during burning are affected by amount of fuels consumed, soil moisture, and soil texture. Lethal temperatures are very likely to be produced under burned slash piles and heavy canopy litter to some depth. Burning herbaceous vegetation and broadcast slash is not likely to increase soil temperatures substantially. These increased soil temperatures under heavy fuels most likely kill soil organisms and propagules.

Our review of the literature suggests that to minimize negative impacts of burning on the soil resource, if slash is to be burned at all, it should be broadcast in interspaces before being burned. Although some nutrients are lost to volatilization, this practice should result in moderate increases in nutrient availability and in short term productivity. However, other impacts of broadcast burning (such as possible reductions in understory production and tree seedling establishment) should be evaluated also.

Because burning unharvested pinyon-juniper is usually done under severe conditions (similar to a wildfire), soil heating and potentially negative impacts on soils are likely to occur under current fuel loadings.

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Fire Effects in Southwestern Chaparral and Pinyon-Juniper Vegetation

Rex D. Pieper and Roger D. Wittie[1]

Abstract—Periodic wildfires kept woodland communities relatively open, but fire suppression techniques encourage denser pinyon-juniper stands. In areas where prescribed fire has been used to open such stands and increase understory productivity, secondary successional patterns are similar. Herbage production often increases following burning but productivity declines as the woodland is reconstituted.

Fire in pinyon-juniper and southwestern chaparral communities is of interest for at least two reasons: the historical role of fire, especially with respect to maintaining grassland vegetation free from encroachment of trees and shrubs; and the use of prescribed fire as a management tool in these communities to increase herbage production for livestock, improve wildlife habitat, and decrease fuel accumulation for reduction of wildfires.

There is ample evidence that the density of trees within stands has increased as well as the total area occupied by woodlands (Pieper 1977, Pieper et al. 1987, Little 1977, Springfield 1976, West et al. 1975 and West 1984). The argument that frequent fires prevented expansion of pinyon-juniper woodland into grassland ranges is related to fire frequency and the efficiency of fire control programs which have reduced the areas burned annually by wildfire. The hypothesis is that periodic fires burned these grasslands at an interval frequent enough to kill any tree seedlings which had become established. When fire control procedures became effective these seedlings were no longer killed by fire and they became established in the grassland which was eventually converted to a woodland or savanna. Coupled with this reduced fire impact was livestock grazing which may have reduced the fuels necessary to carry a fire. Evidence to test this hypothesis is difficult to obtain since stands which now burn are characterized by older and larger trees which are much more resistant to burning.

The role of fire in chaparral communities is obviously different from that of the pinyon-juniper communities. Many shrubs of the chaparral are crown sprouters and are fairly resistant to fire. In addition, chaparral islands in grassland may represent relicts of continuous stands rather than encroachments into grassland vegetation (Cable 1975).

The objective of this paper is to review the historical role of fire in these pinyon-juniper and chaparral communities, to review the ecological influence of fire and the possibility of using prescribed fire to meet certain management objectives. These topics are not mutually exclusive, but will involve some overlap.

Historical Role of Fire

Fire has been a frequent phenomenon in both pinyon-juniper and chaparral vegetation (Mueggler 1976). However, evidence concerning the frequency and extent of fires prior to the advent of European exploration and settlement is scarce. Young and Evans (1981) studied fire scars on western juniper (*Juniperus occidentalis*) trees in low sagebrush communities in Lassen County California. These scars revealed large fires from 1640-1650, and 1830-1840 (fig. 1). However, scars on another tree indicated fires in 1770, 1780, 1790, 1830, and 1850. Use of these techniques to determine fire frequencies is imprecise (Young and Evans 1981). Perhaps fire histories from these western juniper stands can be extrapolated to similar situations for pinyon-juniper vegetation in the Southwest.

Wright et al. (1979) stated that fire frequencies of 10 to 30 years were one factor in restricting pinyon-juniper invasion into grassland. How-

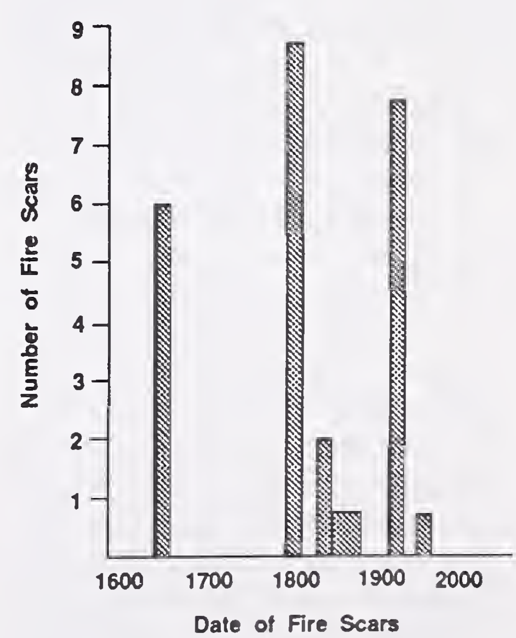

Figure 1.—Number of fire scars on singleseed juniper trees in Lassen County, California (Young and Evans 1981).

[1]Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).

[2]Pieper is Professor of Range Science, and Roger D. Wittie is Graduate Assistant, Dept. of Animal and Range Sciences, New Mexico State University, Las Cruces, New Mexico 88003.

ever, the only source for this fire interval in the Southwest was an earlier article by Leopold (1924) who also relied on fire scars to determine burning interval.

Information on fire history of chaparral vegetation is even more limited than that for the pinyon-juniper type. Cable (1975) stated that chaparral vegetation is not a fire climax although fire is a common occurrence. Some of his observations indicate that a few chaparral stands had not been burned for as long as 100 years. Another area had not been burned for at least 72 years (Pase 1972). Apparently many chaparral areas will not have sufficient fuel to carry a fire at intervals more frequent than 20 years. Chance occurrence of lightning storms and sufficient fuel to carry a fire are apparently two factors controlling fire frequency in chaparral communities.

Pinyon-juniper and chaparral both fit the fire climate defined by Fosburg and Furman (1973). These authors identified four distinct fire types in Arizona. The Central Basin included most of the chaparral type while the Central Basin and Grand Canyon-Mogollon Rim included most of the pinyon-juniper vegetation. Several fire types in New Mexico would include pinyon-juniper vegetation (e.g. Western Mountains, Sacramento Mountains, Northwest Plateau, San Juan Mountains, and Sangre De Cristo Mountains).

After burning occurs, it is difficult to prove the thesis that periodic burning prevented encroachment of pinyon-juniper and chaparral vegetation into grasslands. Several other factors may also be influential in this process. West and Van Pelt (1987) have traced some of these influences and resulting changes in Great Basin pinyon-juniper vegetation (fig. 2). These factors include depletion of understory vegetation through grazing, increase in annual herbaceous vegetation represented by species such as cheatgrass *(Bromus tectorum)*, reduction of woody overstory through post and fuelwood harvesting, etc. Many of these same factors probably influenced pinyon-juniper vegetation in the Southwest.

Fire also played a role in intercommunity cycles in presettlement time leading to alternations between savanna and dense woodland (fig. 3). This model indicates that present vegetation may be only a phase in a cyclic process (West and Van Pelt 1987).

Ecological Responses to Fire

The effect of fire on trees in pinyon-juniper vegetation is influ-

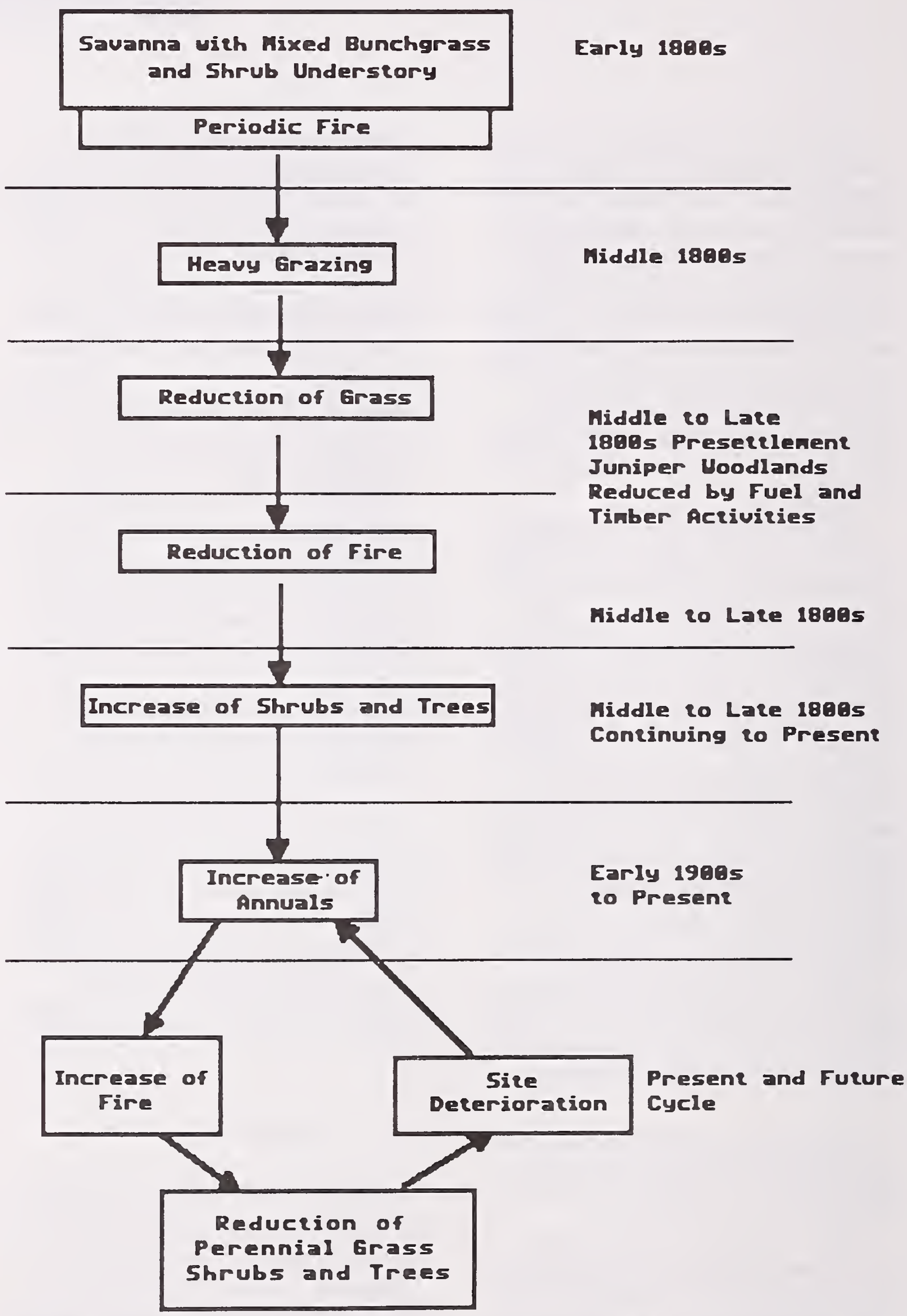

Figure 2.—Historical summary of events leading to current conditions in piñon-juniper woodlands in the Great Basin (from West and Van Pelt 1987).

enced by several factors including size of trees, amount of herbaceous fuel, wind speed, air temperature, stand density, vertical and horizontal fuel distribution and season (Bruner and Klebenow 1979, Wright et al.

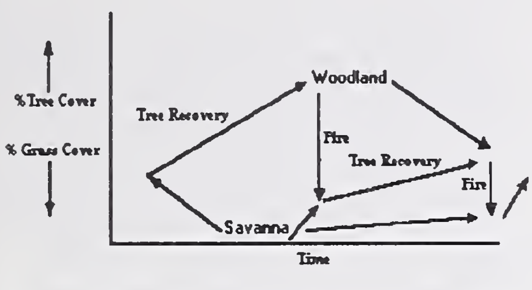

Figure 3.—Changes in pinyon-juniper vegetation and the role of fire in thickening toward woodland and thinning toward savanna (West and Van Pelt 1984).

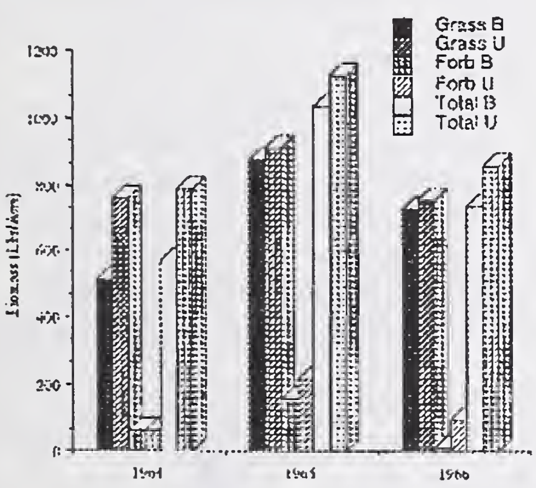

Figure 4.—Herbage production following burning in New Mexico (Dwyer and Pieper 1967).

1979). Dwyer and Pieper (1967) reported that a rapidly moving wildfire killed nearly 1/4 of singleseed juniper *(Juniperus monosperma)* trees and only 14% of Rocky Mountain pinyon *(Pinus edulis)* trees in southcentral New Mexico. However all trees under 4 ft. tall were killed by the fire. Jameson (1962) also reported that from 70 to 100% of singleseed juniper trees under 4 ft. tall were killed by spring fires in Northern Arizona. Mortality of trees 5 to 6 ft. tall was only 30 to 40%. Larger trees with accumulation of Russian thistle fuel under the canopy exhibited mortality of 60 to 90%. Canopy morphology also plays a role in susceptibility of trees to fire. Those trees with the canopy developed close to the ground are much more likely to have the crown catch on fire from fine fuels on the soil surface than those trees which have an elevated canopy. These morphological differences are often related to age of the tree, fire history and species. Pinyon would likely dominate stands under a history of repeated fires.

The effectiveness of wildfire in restricting spread of pinyon and juniper trees into grassland vegetation types depends on fire frequency and intensity of the fire. Information on early growth rate of pinyon and juniper seedlings is lacking. The time required for these seedlings to reach 4 ft. in height appears to be critical. If severe fires occurred at intervals more often than the time required to reach a height of 4 ft., then fires could be effective in preventing encroachment of trees into grassland. Limited data on growth of both pinyon pine and juniper species suggest that many trees less than 4 ft. tall are over 10 years of age (Clendenen 1979 and Fowler et al. 1984). Chojnacky (1987) and Smith and Shuler (1987) presented graphs showing age:height relationships for pinyon and juniper trees, but few determinations were made for trees less than 6 ft. tall which were between 20 and 100 years old. Howell (1940) found that Rocky Mountain juniper *(Juniperus scopulorum)* seedlings were only 1 ft. tall after 8 years. Pinyon pine seedlings may be only 2 in. tall at two years of age and only 3-4 ft. tall at 25 years of age (Fowells 1965). These limited data suggest that fire frequencies were frequent enough to limit tree seedling establishment in grassland under some circumstances.

The initial effects of fire on herbaceous vegetation is a reduction in biomass for 1 or 2 years following the fire. Usually herbaceous vegetation recovers to preburn levels of biomass within a year or two following the fire (Dwyer and Pieper 1967, Jameson 1962) (fig. 4). Often there is an increase in abundance of annuals immediately following the fire.

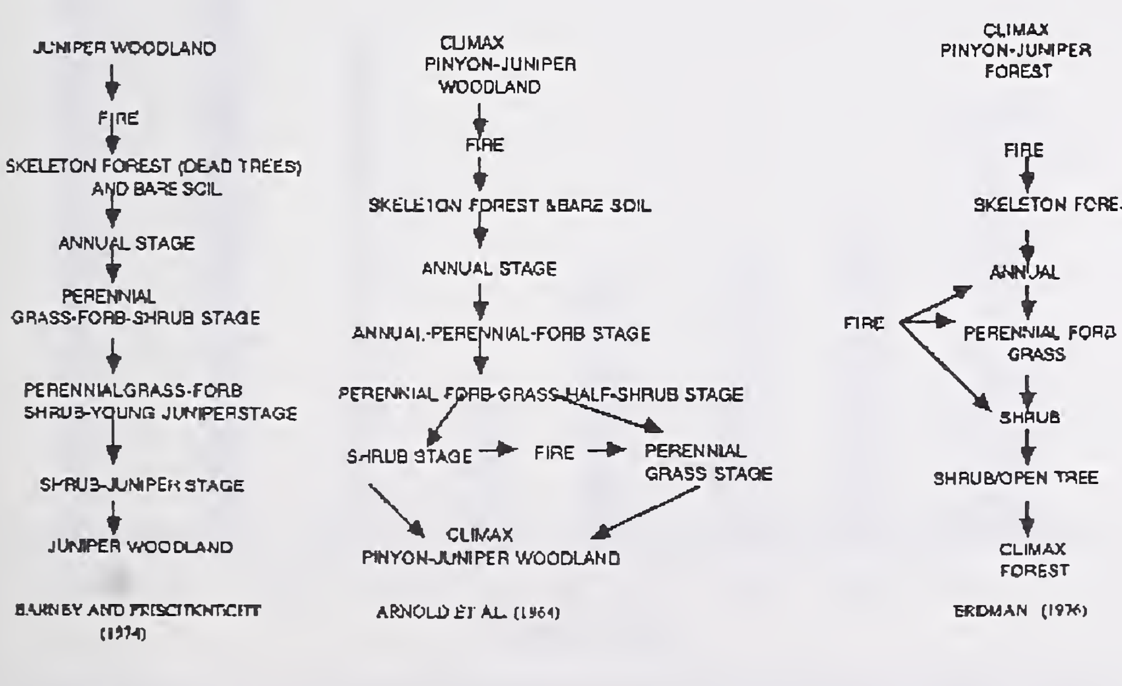

Figure 5.—Successional pattern following fire in piñon-juniper vegetation at three locations.

Successional Patterns

Successional patterns following fire are remarkably similar for pinyon-juniper stands in Arizona, Colorado, and Utah (fig. 5). If the fire is severe enough to burn the tree overstory in addition to the herbaceous and shrubby understory, a skeleton of trees are left with considerable bare soil. The first vegetation becoming established following the

fire consists of annual herbs. In Nevada, annual herbs in early successional stages included coyote tobacco *(Nicotiana attenuata)*, flixweed tansy mustard *(Descurainia sophia)* and goosefoot *(Chenopodium album)*. In Utah, cheatgrass brome *(Bromus tectorum)*, Russian thistle *(Salsola pestifer)*, sunflower *(Helianthus annuus)*, pale alyssum *(Alyssum alyssoides)*, flixweed tansy mustard, and coyote tobacco were common components of the vegetation during early stages of succession. Successional patterns can be altered by fire intensity, season, post burn environment and past history of use.

The annual communities are gradually replaced by mixed annual and perennial species and eventually primarily by perennial species (fig. 5). The three models presented in figure 5 show a pronounced shrub stage before tree regeneration begins. However, in some cases this shrub stage may be reduced or absent (Schott and Pieper 1986, 1987).

The entire sequence from skeleton forest to climax pinyon-juniper woodland varies depending on climatic factors and soil characteristics. Erdman (1970) showed a time frame of 300 years to complete the sequence for his model in Colorado (fig. 5). Barney and Frischknecht (1974) found that the maximum number of pinyon and juniper trees in their study in Utah occurred 86 years following the fire (fig. 6). The density of trees declined slightly from 86 to 100 years.

Although these diagrams indicate that succession is relatively uniform from place to place, there are exceptions. Everett and Ward (1984) described multiple successional pathways and multiple entry points where fire could play a controlling role. These authors stated "Initial and relay floristics successional models are both operative in pinyon-juniper succession. Original preburn species rapidly reappear on the burns, perennial species outnumber annuals, and the species turnover rate rapidly declines (initial floristics); but relay floristics (migration) determines the character of later stages, as we did not find singleleaf pinyon seedlings." In New Mexico grass forb and shrub stages were poorly represented on lithic hapustoll soils with extensive rock outcrops (Schott and Pieper 1986, 1987).

Successional patterns are somewhat truncated in chaparral communities compared to that in pinyon-juniper stands (Cable 1975). The forb stage in secondary succession reaches a peak during the second or third year following the fire. The grass stage reaches a peak from the fifth to seventh year following the fire. Most shrubs are sprouters and reestablish their canopies relatively rapidly after the fire (Pase and Lindenmuth 1971) (fig. 7). Hibbert et al. (1971) reported that more than 11 years was required for the shrub cover to reach preburn conditions. Pase and Lindenmuth reported that only desert ceanothus *(Ceanothus greggii)* and manzanita *(Arctostaphylos pungens)* did not sprout following a fire on the Sierra Ancha Experimental Forest in Central Arizona.

Use of Prescribed Burning

Prescribed burning is often used in pinyon-juniper vegetation to reduce overstory cover and increase understory production (Bunting 1987) or to reduce fuel levels for wildfire prevention (Kallander 1969). Prescribed burning has been used in several ways: (1) burning slash and debris left from other control methods, (2) burning individual trees, (3) burning grassland or sagebrush/grassland to kill invading trees, and (4) broadcast burning mature trees (Blackburn and Bruner 1975).

Recommendations for broadcast prescribed burning in pinyon-juniper vegetation varies with characteristics of the vegetation (Wright et al. 1979). For dense stands, Zwolinski and Ehrenreich (1967) recommended bulldozing a fire line around the area

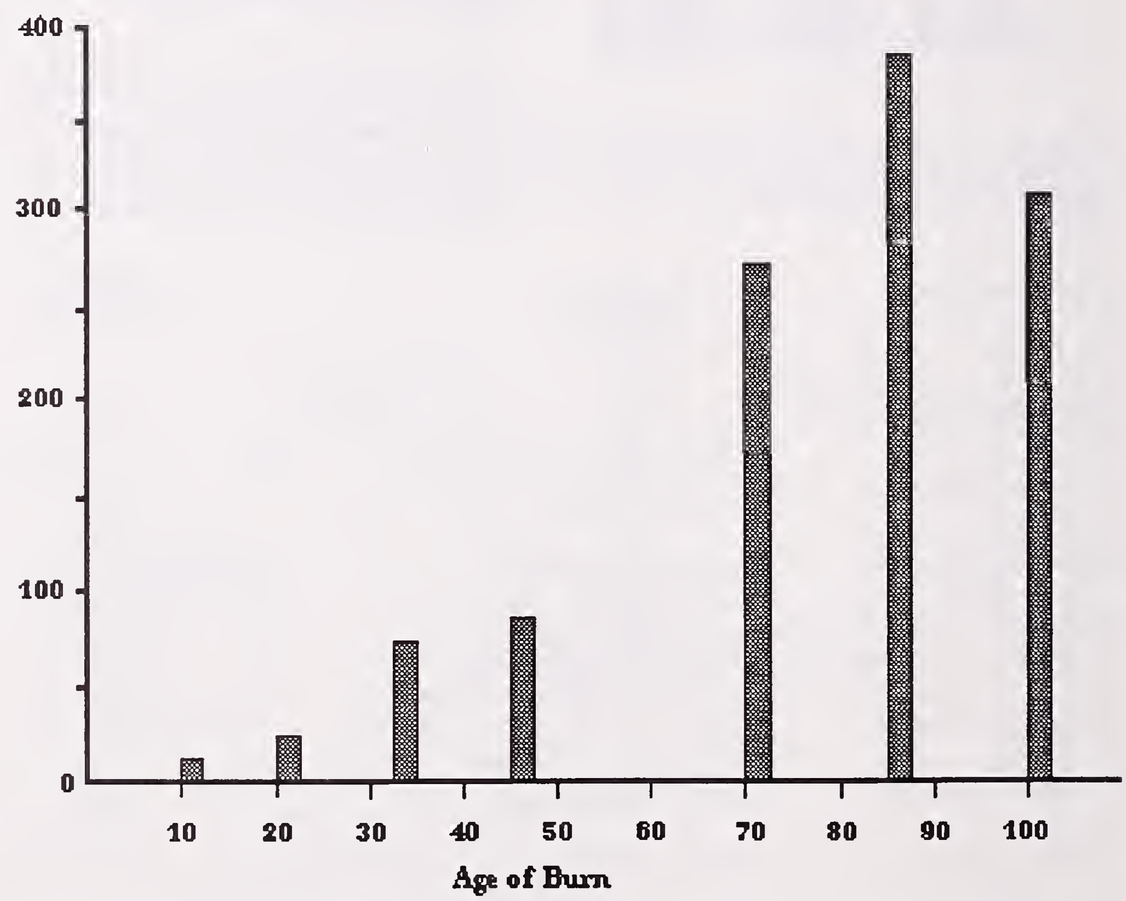

Figure 6.—Density of trees (Utah and singleleaf piñon) following fires in Utah (modified from Barney and Frischknecht 1974).

to be burned and igniting the fuel during peak burning conditions in late June or early July in Arizona. Prescribed burning recommendations on the Hualapai Reservation in Arizona call for burning during the hottest and driest conditions possible during the year which is in late June before the summer rainy season (Despain 1987). Most burns have been conducted between June 20 and July 10. Other recommendations listed by Despain (1987) for this Reservation include the following:

- Fires should be ignited between noon and 3:00 p.m.
- Air temperatures must be in the high 80s or higher
- Wind velocity should be from 15-20 mph and steady
- Relative humidity should be less than 8%

Burning under these extreme conditions increases the possibility of the fire escaping the desired area to be burned. Maximum precautions are necessary and these will add to the cost of the burn. Often closed stands of pinyon-juniper have little understory and are difficult to burn (Bruner and Klebenow 1979, Aro 1971, Blackburn and Tueller 1970, Wright et al. 1979). In these cases it may be necessary to use some other method to push the trees over where they are susceptible to burning.

Prescribed burning has been a part of the overall management plan for the Hualapai Indian Reservation in northwestern Arizona (Despain 1987, McNichols 1987). Approximately 22,000 acres have been burned with an additional 100,000 acres planned for burning (McNichols 1987).

Individual tree burning is desirable under some conditions where broadcast burning is not practicable (Jameson 1966, Springfield 1976, Wright et al. 1979). Propane or oil-burning torches can be used to ignite the trees (Jameson 1966). However, the method is not practical for large trees (10 ft. or taller) (Springfield 1976).

Baldwin (1968) has described procedures for prescribed burning of chaparral vegetation on the Tonto National Forest in Arizona. For best results burning should be done when fire conditions are extreme and precaution is necessary to keep the fire under control. The fire is ignited on the ridge tops and progressively down the slopes until it is safe to start fires on the bottom of the slopes for the main fire to burn rapidly uphill. Follow-up management including seeding with grass species and herbicidal treatment 18 months following the burn to control sprouts and seedlings of the shrubby species is used on the Tonto National Forest. Pase and Lindenmuth (1971) described the use of mixtures of 2,4-D and 2,4,5-T as desiccants before burning chaparral vegetation. They recommended that the moisture content of desiccated leaves be less than 15% and nondesiccated leaves more than 85%. Cable (1975) listed the following conditions for favorable burning of chaparral vegetation:

- vegetation must be dormant or nearly dormant
- wind should not exceed 4 miles per hour
- relative humidity should be between about 14 and 35%
- fuel stick moisture should be between about 8 and 18%

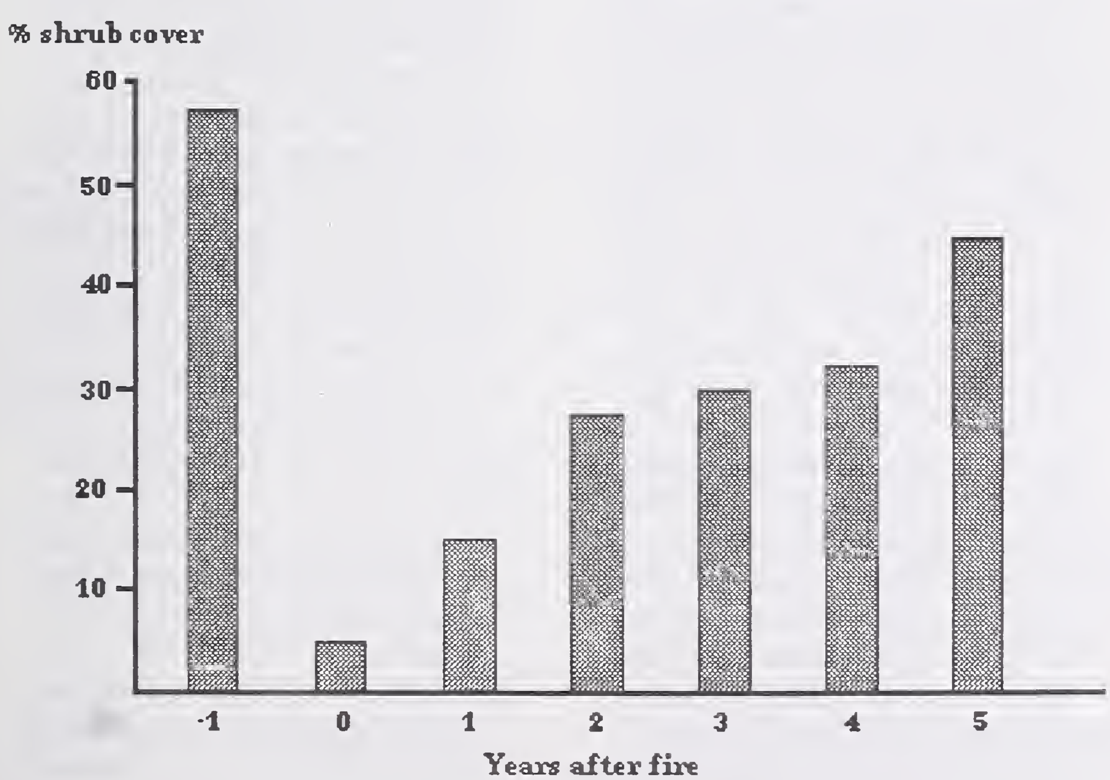

Figure 7.—Cover of shrubby species following fire in chaparral community in Arizona (Pase and Lindemuth 1971).

Summary And Conclusions

Wildfires have played an important role in the development of pinyon-juniper and chaparral vegetation in the Southwest. Because of the relatively slow growth rate of both pinyon and juniper trees, fire frequencies were apparently frequent enough to restrict encroachment of tree seedlings into grasslands. However, the role of fire in these woodlands has changed with the advent of livestock grazing which reduced fuel loads and efficient fire control programs which reduced the extent and frequency of fires in these communi-

ties. Now prescribed fires are being used as tools to reduce tree density and increase understory biomass for improved habitat for livestock and game. Prescribed burning techniques have been developed to meet management objectives for different areas.

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Increasing Habitat Diversity in Southwestern Forests and Woodlands via Prescribed Fire[1]

Abstract.--Historically, prescribed fire has been used as a management tool primarily to create habitat diversity in all upper elevation (>5,000-ft) vegetation types. Research is needed on the feasibility of using this tool in riparian-stream areas and in creating "habitat mosaics."

Kieth E. Severson and John N. Rinne[2]

Fire occurs in most, if not all, of the vegetation types in the Southwest, and, in concert with climate and soils, is responsible for development of existing biotic communities in Arizona (Pase and Granfelt 1977). The relative importance of fire in shaping these communities varies. Moist areas such as some riparian habitats, wet meadows, and upper elevations of spruce-fir are often too wet to burn; hence, fires in these areas occur relatively infrequently. In arid environments, such as deserts, lack of fuel precludes regular burning. Fires did influence vegetation and associated animals wherever fuels accumulated and where there were seasonal dry periods. There have been several recent and thorough reviews related to fire and wildlife; namely, on birds and mammals (Bendell 1974), on fauna (Lyon et al. 1978), on wildlife (Peek 1986, Wright and Bailey 1982). The reviews by Lyon et al. (1978) and Wright and Bailey (1982) included discussions on effects of fire on stream fauna.

We will complement the information available in these reviews with information specific to the biotic communities of the Southwest. Bock and Bock (these proceedings) discuss effects of fire on wildlife in lower elevation communities; we treat those communities beginning with Great Basin Conifer Woodlands and continuing upward. We also discuss effects of fire on fishes and other aquatic organisms, particularly trouts inhabiting higher elevation streams.

Wildlife and Fire

Fire influences wild animals in two general ways; by killing directly and by altering their habitat. This paper emphasizes fire effects on habitat. However, concern is often expressed regarding the direct effects of fire on wild animals--death resulting from exposure to heat or smoke. The reviews cited discuss these effects and generally conclude that such mortality is uncommon; most animals have the ability to escape by moving or burrowing. Where mortality does occur, areas are rapidly recolonized by immigration or by an animal's innate ability to reproduce rapidly. Generally the beneficial effects of fire on wildlife, especially in habitats where fire commonly occurs, offset potential losses.

While wildfires will be mentioned occasionally, our primary interest is in the use of prescribed fire; that is, fire as a management tool designed to be used under a specific set of conditions to burn a predetermined area for a specified purpose or purposes. Contemporary wildfires tend to burn during hot, dry periods, in areas of accumulated plant debris, and tend to be severe and difficult to control. Although Peek (1986) did mention several examples where wildfire benefitted native ungulates, he and others generally agree that many benefits resulting from earlier wildfires have been fortuitous and were likely accompanied by other less desirable effects.

Prescribed fire can improve habitat for wild animals primarily by increasing diversity in the shrub and herb layers. Fire can alter both the relative abundance of forage plants and their nutritive contents. It also can be used to augment diversity in habitat structure by breaking up homogeneous cover types. We will discuss three strategies for increasing habitat diversity via prescribed fire--altering forage composition, nutritive content of forages, and habitat structure--in the major woodland and forest types in the Southwest: Great Basin Conifer Woodlands, Montane Conifer Forests (including the ponderosa pine *(Pinus ponderosa)* and mixed conifer series), and the Rocky Mountain Subalpine Forest. Descriptions of these forest types are provided by Brown (1982).

Great Basin Conifer Woodlands

Great Basin Conifer woodlands (hereafter called pinyon-juniper woodlands) are distributed through-

[1]Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).

[2]Range Scientist and Fisheries Biologist, Rocky Mountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287-1304.

out the northern two-thirds of Arizona and nearly all of New Mexico and should not be confused with Madrean Evergreen Woodlands which are restricted to relatively smaller areas in southeastern Arizona and southwestern New Mexico. The role of fire in development of pinyon-juniper woodlands has been relatively well defined (Wright et al. 1979); fire, drought, and competition were the key factors limiting distribution of this type before the introduction of livestock (Wright and Bailey 1982). Cessation of burning in pinyon-juniper resulted from active suppression programs and the removal of fire-carrying herbaceous material by livestock, likely causing the spread and increased densities of pinyons and junipers in these stands (Wright et al. 1979). Studies on effects of fire have examined both wildfires and prescribed burns, but most have dealt with increasing forage production for livestock; less emphasis has been given wildlife habitat.

Wildlife prescriptions for controlled burns south of the Mogollon Rim emphasize rejuvenating decadent browse stands.[3] These shrub species, the same ones present in Interior Chaparral, readily sprout from roots and crowns; hence, objectives and results are similar to those discussed by Bock and Bock (these proceedings). Above the Rim, on the Coconino and Kaibab Plateaus, most of the shrubs associated with pinyons and junipers are weak or nonsprouting species such as big sagebrush *(Artemisia tridentata)* and cliffrose *(Cowania mexicana)*. Prescribed burns have significantly increased herbage production which has resulted in heavy use by mule deer *(Odocoileus hemionus)* during fall and winter in areas where snow cover was not a problem (McCulloch 1969). To prevent shrub mortality where snow accumulated, McCulloch (1979) recommended using other methods, such as bulldozing, to control pinyons and junipers. More recent evidence indicates, however, that prescribed burning may be feasible even under these circumstances. In 1979, a 1,200-acre (528 ha) prescribed burn in pinyon-juniper on the North Kaibab resulted in high shrub mortality. While cliffrose sprouted the following year, the sprouts immediately died back. Now, photographic evidence shows a vigorous stand of cliffrose that apparently resulted from seed.[4]

The primary objective of pinyon-juniper burning on the Kaibab National Forest is to improve wildlife habitat by increasing diversity of vegetation.[4] Younger burns may be dominated by herbaceous species such as western wheatgrass *(Agropyron smithii)*; slightly older burns show greater amounts of shrubs such as cliffrose, and older burns are being reinvaded by pinyons and junipers. It is difficult to predict early plant succession on pinyon-juniper burns because it depends on (1) where the burn leaves the site within the successional framework (e.g., cool or hot fire), (2) an unpredictable response from soil seed reserves, (3) postfire weather, and (4) the availability of plants adapted to postfire conditions. However, diversity of species increases as preburn species return to the site (Everett and Ward 1984). Thus, different-aged burns, juxtaposed among themselves and with unburned areas, maximize diversity. Unburned sites are needed as cover areas for large ungulates (Severson and Medina 1983) as well as habitat for tree nesting birds. Southwestern pinyon-juniper woodlands also function as wintering areas for a multitude of nongame birds. Maintenance of a mature pinyon and juniper overstory for food production (seeds and berries) and cover is important for these wintering populations (Balda and Masters 1980). Trees in closed stands of pinyon-juniper with no significant understory can be very difficult to kill because such stands do not carry fires well (Wright and Bailey 1982). The Kaibab National Forest strives for diversity in these stands by walking through them with drip torches, igniting only those areas that have the proper overstory-understory conditions for small (2-4 acre [0.88-1.6 ha]) spot burns.[4]

No information is available on effects of burning on bird habitat in this type, but studies have been done comparing populations on chained areas with those in untreated pinyon-juniper stands. Total number of birds and number of species were consistently higher on untreated plots (O'Meara et al. 1981, Sedgwick and Ryder 1987). In the chained plots, where overstory foliage was markedly reduced, the species dependent on foliage and live trees for nesting or foraging declined (e.g, black-throated gray warbler *[Dendroica nigrescens]*, solitary vireo *[Vireo solitarius]*, white-breasted nuthatch *[Sitta carolinensis]*, and gray flycatcher *[Empidonax wrightii])*. Removing snags eliminated cavity nesters such as the hairy woodpecker *(Picoides villosus)*. Only two species, chipping sparrow *(Spizella passerina)* and rock wren *(Salpinctes obsoletus)*, consistently nested in the chained areas, although other species frequently foraged there.

Although not stated, the sizes of chained areas were likely quite large. Sedgwick and Ryder (1987), for example, recommended that chainings should be kept less than 600 ft (200 m) wide, which, based on their results, imply those in their study area were larger. If treated areas are large, whether chained or burned, the impacts on birds will be significant. Declines in tree foragers/nesters and cavity dwellers are not compensated for by increases in ground/shrub foragers/nesters. Similar responses probably would

[3]Warrick D.; Fager, L., USDA Forest Service (personal communication).

[4]Zarlingo, V, USDA Forest Service (personal communication).

not occur if treated areas were small rather than large even if the total area treated was the same size. While treating small areas may result in a slight decrease in numbers of tree nesters, it should be compensated for by an increase in ground nesters. Species diversity would likely increase.

Another way to manage overstory in closed stands is to integrate burning with fuelwood harvesting or, in certain circumstances, mechanical methods such as bulldozing and cabling (chaining). These techniques create slash and disposal and can present a dilemma because slash, especially juniper, tends to last a long time. Some slash retention can be beneficial; too much can cause problems. Excessive accumulations can interfere with elk *(Cervus elaphus)* and livestock distributions and can also reduce herbage production (Severson and Medina 1983). Conversely, in certain situations, slash may afford benefits. Limited information suggests that slash accumulations may provide mule deer, which tend to use areas with slash, with security areas to escape adverse behavioral interactions that might occur between deer and elk or cattle (Reynolds 1966, Severson and Medina 1983). Long sighting distances can also be broken up by piling or windrowing slash, thus providing wildlife a form of security (Thomas et al. 1979). Scattered slash can favor developing plants by providing a favorable microclimate and protection from grazing herbivores. Slash has often been piled or windrowed and then burned. This practice has been criticized because the extreme heat generated under these circumstances severely alters soil conditions; hence, subsequent community development starts at an earlier successional stage and proceeds at a slower rate. There are, however, certain advantages to removing some of the slash--diversity and numbers of certain animals can be increased. Severson (1986a) studied rodent populations on pinyon-juniper woodlands subjected to four treatments; bulldozed/piled/burned, bulldozed only (slash not piled), thinned (slash left in place), and undisturbed woodlands. Rodent numbers were higher on all treatments compared with undisturbed woodlands, but species composition differed. Plots that had been bulldozed/piled/burned contained greater proportions of "grassland" species such as white-footed mice *(Peromyscus leucopus)*, southern grasshopper mice *(Onychomys torridus)*, and Ords kangaroo rat *(Dipodomys ordii)*. Plots with slash, thinned and bulldozed only, held proportionately more woodrats (*Neotomys* spp.).

The spots on which slash had been burned contained a different flora than the surrounding area 20 years after treatment. The additional floristic diversity provided by these younger seres gives foraging animals a greater variety of foods to choose from and also forms a different cover type for smaller animals. Because of the severity of the treatment, such areas should be small (400 ft^2 or less [37 m^2]) and scattered. The total area treated should remain small enough so the overall productivity and hydrologic capabilities of the watershed are not impaired.

Management Implications

Prescribed fire is a useful tool for managing pinyon-juniper habitats. It can be used to create mosaics within the canopy and within the resultant herbaceous fields. While fire protection and overgrazing have contributed to the creation of closed canopies that are difficult to treat with fire, it can still be used in conjunction with other treatments, such as fuelwood harvesting, to achieve desired results. Optimum size of burned areas and their arrangement relative to other treated and nontreated areas will depend on management goals. Based on work with small birds (Sedgwick and Ryder 1987) and mule deer (Lechenby 1977, McCulloch 1979), the optimum width of treated areas should not exceed 660 ft (201 m); or, if regularly shaped, they should not exceed 10 acres (4 ha).

Montane Conifer Forests (Ponderosa Pine Series)

Probably more attention has been given to fire in the Ponderosa Pine Series than in any other biotic community in Arizona and New Mexico. Effects of historical and recent wildfire and uses of prescribed burning have been given in other reviews (Severson and Medina 1983, Wright and Bailey 1982). Both in research and application, most emphasis has been on the use of prescribed fire as a tool to remove excessive fuel accumulations. Primary benefits to wildlife have been increases in understory (forage) production and temporary increases in nutritive content of forage plants. Ancillary uses of fire have been to increase growth of shrub species, especially buckbrush *(Ceanothus fendleri)*, and to examine fire's value for increasing populations of certain threatened/endangered/sensitive plant species.[5]

Understory production, while perhaps variable the first 2 years, does increase in response to burning in ponderosa pine (Andariese and Covington 1986, Harris and Covington 1983, Oswald and Covington 1984, Severson and Medina 1983). Results have been attributed to reductions in the number of live, competing trees, amount of litter, and depth of duff. Using prescribed fire to increase the nutritive value of understory forages, however, has some complications.

Perhaps the first thing that piqued manager's interest in the relationship between fire and wild animals was the way that fire (or the results of fire) altered the distribution of animals, particularly ungulates. Ani-

[5] *Goodwin, G., USDA Forest Service (personal communication).*

mals are often attracted to a burned area immediately after a fire, sometimes gathering on the blackened surface. Komarek (1969) noted instances in Africa where native animals were observed "nibbling" on recent burned areas, presumably consuming ash for nutrients. The strongest attraction, however, is provided by the greening stage after the burn. Little quantitative information exists on animal response to the initial green flush of vegetation, but most of us have observed it and, like Komarek, likely attributed it to "fresh, nutritious herbage."

The first point to consider when using prescribed burns to achieve this "nutrient flush" is that such responses are short-lived. Severson and Medina (1983) reviewed eight papers that treated this topic and all indicated that the nutrient content of plants growing on burned areas was higher than that of plants growing on preburn or control areas. Six of the eight studies noted, however, that nutrient contents of forage plants tended to revert to control or preburn status in 2 years or less. A study in Arizona ponderosa pine revealed a typical pattern. Pearson et al. (1972) found that crude protein, phosphorus, and in vitro digestible dry matter were higher in forages from burned areas the first growing season. The increases in digestible dry matter and phosphorus lasted through the second, but increases in protein lasted only during the first. No differences were seen by the end of the second growing season.

The other point warranting attention is that relatively "warm" fires apparently have to be used to achieve these goals. Stark (1980) found that light fires (soil surface temperatures <150°F [66°C]) were not suitable for improving the quantity or quality of browse plants. She recommended that soil surface temperatures of 572°F (300°C) or higher would be necessary to obtain significant increases in foliar nutrient concentrations. However, if soil temperatures are allowed to become too hot, nutrient losses will occur including nitrogen, a major nutrient and the key constituent in protein. DeBano and Conrad (1978) reported, from California chaparral, that in a prescribed burn where soil surface temperatures averaged about 715°F (380°C), nitrogen loss represented about 11% of that in plants, litter, and the upper 4 inches (10 cm) of soil. White et al. (1973) recommended that heating of ponderosa pine litter be kept below 572°F (300°C) to minimize losses of total nitrogen. We would infer that in prescribed burns intended to increase nutritive contents of forages while retaining as much total N as possible in the system, soil surface temperatures should be kept around 550 to 600°F (288 to 316°C).

It is feasible and practical to burn areas at intervals to increase the nutritional value of plants. This could be done with prescriptions for fuel reduction provided a relatively short treatment rotation could be realized. Creating a mosaic of understories with different nutritional attributes (e.g., "nutritional diversity") could be achieved by developing prescriptions that burned different patches every year and reburning at intervals not to exceed 4 years (Covington and Sackett 1986). Long-term influences of short rotation burning are unknown; presumably, however, there would be no detrimental effects. Ponderosa pine ecosystems apparently evolved under a regimen of frequent burning. Dieterich (1980) found an average interval of one fire every 4.9 years over a 336-year period in northern Arizona ponderosa pine. During one 126-year period fires burned at 2.5-year intervals, and over one 15-year period there was one fire every 1.25 years. Variations in fire intensity and in burning over time and space, and incorporating burning patterns with timber harvest, would likely provide needed diversity in habitats, forage species, and nutritional content of those species.

Two precautionary notes: The effects of prescribed burning for fuel reduction can differ between situations where it is being used initially; that is, where fire has been excluded for a long time, and where it has been used on a regular basis. Second, not all studies have documented a nutritional response to regular burning. Wood (1988), working in southeastern longleaf pine *(Pinus palustris)*, did not detect any differences in weights of dry matter and nutrients in either total forage or known palatable forage in burned vs control plots. He concluded that changes in forage quality were small and lasted only a few months.

Little information is available on the effects these cool, repeated burns could have on smaller fauna. In the initial stages of burning, small mammals may decrease as forest floor residues are reduced. Once the understory vegetation has responded these animals may increase because of additional foliage cover and seeds. Bird numbers may not be affected greatly although an increase in ground foragers and nesters may result from the increased understory. While wildfires may create snags--potential habitats for cavity nesting birds--by killing trees, prescribed ground fires may destroy them. A single prescribed burn described as "moderately intense" burned nearly half of all ponderosa pine snags with d.b.h. greater than 6 inches (15.2 cm) (Horton and Mannan 1988). Those most susceptible had large amounts of loose, relatively undecayed, woody debris at the base. Hence, some protection will have to be afforded snags under a repeated burning program.

Wild animals also respond to habitat conditions created by intense fires, i.e., wildfires or any fire that removes the pine overstory, but the reasons are apparently different. Although the "nutrient flush" only lasts 1 to 2 years, animals continue to fa-

vor areas that were more severely burned. A review of six papers (Severson and Medina 1983) gave these reasons for the long-term attractiveness of burned areas to animals:

1. Increased habitat diversity or "edge."

2. Increased production of preferred forage.

3. Increased forage diversity.

4. A combination of the above also including the cover provided by the dead, standing trees.

Small bird responses to severe treatment (wildfire) in a California mixed pine type (*Pinus jeffreyi, P. ponderosa,* and *P. washoensis*) demonstrated the potential effectiveness of habitat diversity, spatially and temporally, on breeding populations. Fire was intense enough to destroy the overstory but left standing dead boles and occasional unburned pockets of mature timber.

During the early stages of recovery (6-8 yr postburn) 9 species of birds were unique to the burned area, 6 to the unburned forest, and 17 species were found in both habitats. Species that foraged among needles of living conifers were more common in unburned forest while those characteristic of low brush and open ground predominated on the burn (Bock and Lynch 1970). Continued monitoring of the site revealed, 25 years postburn, that bird numbers changed on the burned site but remained relatively stable in unburned forest.

Differences were attributed to changes in the vegetation on the burned area: (1) standing dead trees which served as foraging and nesting sites declined to about 20% of the postfire density, (2) shrub cover increased twofold, and (3) and density of live overstory trees increased by 50%. As a result, birds dependent on snags decreased, those that nested or fed in shrubs increased, and those that nested or fed in the canopies of overstory trees increased. Changes in bird numbers were related to the successional pattern of vegetation development (Raphael et al. 1987).

Management Implications

There is evidence that fire can be used in ponderosa pine to benefit small birds as well as the larger herbivores. Relatively cool fires, burning 25 to 33% of the forest floor in irregular patches under sapling, pole, and sawtimber stands each year, in a 3- to 4-year rotation, will create understory mosaics that vary in production, composition, and nutritional value. Hotter, more severe fires can be used to alter the overstory and initiate earlier seral stages, but such sites should be limited in size and well distributed.

There is some indication that small spot fires of high intensity were a normal component of southwestern pine forests.

White (1985) suggested that aggregations of pine seedlings were established when 1 or 2 large trees died and the resultant fuel burned, leaving an intensely burned spot which, in conjunction with adequate seed production and favorable moisture conditions in the spring, created ideal seedbed conditions. Southwestern ponderosa pine forests, therefore, appear adapted to both kinds of fire: extensive, cool fires that reduced fuel accumulations and small, intense, spot fires that created seedbed conditions.

Fire, by itself, may not be an appropriate tool for type conversion (i.e., converting from a type dominated by pine to an early seral stage) on forested summer ranges because of the intense crown fires that would be necessary to achieve the goal. It could, however, be a useful tool to use with timber harvest to achieve such conversions on selected sites.

Mantane Conifer Forest (Mixed Conifer Series) Rocky Mountain Subalpine Forest

We have elected to combine these two biotic communities because their response to fire and the way fire can be used within each to improve wildlife habitat are similar.

The Mixed Conifer Series occurs from 8,000 to 9,500 ft (2,450-2,900 m). Douglas-fir *(Pseudotsuga menzesii)* is the most common species, mixed with spruce (*Picea* spp.) and fir (*Abies* spp.) at upper elevations and ponderosa pine and blue spruce *(Picea pungens)* at the lower limits (Brown 1982). The Rocky Mountain Subalpine Forest type is found between 8,500 and 12,000 ft (2,600 and 3,800 m). Dominant species include Engelmann spruce *(Picea engelmannii)* and subalpine fir *(Abies lasiocarpa)* (Brown 1982).

Wildfire is far less frequent in these types than in ponderosa pine. Data from the White Mountains of Arizona indicate that mixed conifer forests burned on the average every 22 years (Dieterich 1983). Although fires did occur naturally in subalpine forest, they were even less frequent because of the mesic nature of the habitat. When they did happen, generally during a drought year, they tended to be severe.

Quaking aspen *(Populus tremuloides)* is the principal successional species after fire or other forest disturbances in both these communities (Brown 1982). Fire, over the last 150 years, is responsible for the aspen stands that exist throughout the west and for the even-aged structure of many of these stands (DeByle et al. 1987, Jones and DeByle 1985). Fire suppression has resulted in large, overmature stands with aspens often being overtopped by conifers. Rejuvenation of these stands is critical, because only 7% of the estimated 479,000 acres (193,850 ha) of aspen in Arizona and New Mexico is in the seedling and sapling stage; the rest are mature or overmature (Patton

and Jones 1977). A survey of aspen over the West has indicated that without management intervention seral aspen stands will probably be replaced by conifers, and stable ones may become all-aged and less productive (DeByle et al. 1987). Although relatively stable aspen stands have been identified in other areas of the Rocky Mountains (Harniss and Harper 1982), those in the Southwest are seral and will be eventually replaced by conifers, perhaps at a faster rate than those in western Colorado.[5]

Aspen stands have three characteristics that make them important components of wildlife habitat. Two relate directly to diversity. First, the herbaceous understory is more productive and diverse in aspen stands than in adjacent conifer stands. Reynolds (1969) found that an aspen stand produced 6 times the understory of a mixed conifer stand in Arizona. Severson (1982) noted that only moist, grass-dominated meadows produced more forage than did unmanaged aspen stands in the Black Hills. Aspen stands also contain more species in the understory than do ponderosa pine stands. Kranz and Linder (1973) found 54, 49, and 39 species in the understories of aspen, mixed aspen-pine, and pine stands, respectively, in the Black Hills. Second, aspens are the only deciduous trees at higher elevations in the Southwest and contribute significantly to the enhancement of diversity and creation of edge. Lastly, aspen itself is a palatable and nutritious browse consumed by elk, deer, and cattle. This characteristic often creates serious problems in managing aspen stands, as will be discussed later.

Management of aspens is further complicated by the variety of seral stands, each of which provides a slightly different kind of habitat. Pure aspen forests, some with, others without a shrub understory; and mixed aspen-conifer, some with conifers in the overstory, others with conifers in the understory; all provide markedly different habitats for wildlife, especially birds (DeByle 1985). As an example, we can compare habitats for two forest grouse, the ruffed grouse *(Bonasa umbellus)* and the blue grouse *(Dendragapus obscurus).* Ruffed grouse require dense, small aspen growth without conifers for brood habitat and older age classes (large male trees), again without conifers, for winter habitat (Gullion and Svoboda 1972). Male blue grouse, however, select Douglas-fir stands with large aspens present in the overstory (a sere successionally beyond the aspen stage) for spring and summer habitats (Severson 1986b).

The value of aspen forests to wildlife has been thoroughly treated in a review by DeByle (1985) who listed 56 mammals and 77 birds that were found in western aspen stands. Apparently little is known about reptiles and amphibians living in or dependent upon aspen habitats. Two more recent studies, one with birds and the other with elk, have provided additional data. Scott and Crouch (1987) found no difference in density of breeding birds between aspen forests with 25% of the area in clearcut patches of 3 to 20 acres (1.2-8.1 ha) and uncut aspen; however, species diversity was increased. They concluded clearcutting aspen in small blocks on an 80-year rotation would increase "edge," species diversity, and even total number of birds. Canon et al. (1987) compared elk foods and feeding habits among several habitats, including burned and unburned aspen stands and mixtures of aspens and conifers. They found no dietary nutritional differences between burned and unburned aspen stands but did note that time spent feeding was substantially greater on burned aspen sites, probably because preferred forages were consistently available.

The presence of aspens in a forest landscape benefits not only wildlife. The aspen understory also furnishes significant forage for livestock; aspen has potential as an energy source in livestock feed formulations. Aspens provide excellent watershed protection and are aesthetically desirable, both as a foreground and a background vegetation. They can even be managed as fuel breaks because of low ignition rates, low burning index, and lack of ability to carry a crown fire (Severson and Medina 1983).

Patton and Jones (1977) suggested management techniques for aspen stands in Arizona and New Mexico and developed recommendations for three different types of stands: (1) conifer-aspen mixtures, (2) aspen stands with coniferous understories, and (3) aspens with no coniferous understories. Recommendations for all three types involved cutting; prescribed fire was not mentioned. Severson and Medina (1983) expanded on this idea, using the same general breakdown. They concluded, that although cutting may create uneven-aged patches in young and mature stands, it alone may not result in significant sprouting in overmature stands. Intense wildfires have been shown to promote sprouting under these circumstances, however (Patton and Avant 1970). Therefore, clearcutting, followed by broadcast burning of slash may yield better results. A review of several papers indicates that, as aspen stands pass maturity, higher intensity fires may be necessary to stimulate adequate sprouting because increased root temperature, caused by exposure of soil to sunlight, is the cardinal factor in stimulating suckering. In younger stands, increased soil temperature, resulting from clearcutting, and light burning, which creates a blackened surface, also has resulted in sprouting. A moderate- to high-intensity burn creates higher soil temperatures in two ways; directly, by heating the mineral soil, and indirectly, by removing all litter and duff and by creating a blackened surface (Severson and Medina 1983). There is, appar-

ently, no danger of too much heat. Horton and Hopkins (1965) found it impossible to prevent root sprouting by intense burning.

Brown (1985), while not relating specifically to the seral age of aspen stands, suggested the following results (based on work by Bartos and Mueggler (1981) and Horton and Hopkins (1965)) would likely occur under three levels of fire severity:

1. **Low fire intensity, low ground char.**

   Vegetation is partially killed; no more than 1/2 to 2/3 of the aspens are killed; litter and upper duff only are consumed. Response: suckering is patchy and relatively sparse; least effective response to fire.

2. **Moderate to high fire intensity, moderate ground char.**

   All or nearly all aspens are top-killed; patches of duff and charred material remain. Response: suckering is prolific; most effective response to fire.

3. **Moderate to high fire intensity, high ground char.**

   All aspens are top-killed; forest floor is reduced to ash and mineral soil is exposed. Response: suckering is substantial; intermediate between low and moderate severity levels.

Prescribed fire has not been commonly used to regenerate aspens because the aspen forest is difficult to burn (Brown and Simmerman 1986). Also, the understory is often reduced by grazing livestock before these fine fuels can cure and contribute to fire spread (DeByle et al. 1987, Jones and DeByle 1985). As conifers invade, the site becomes more flammable. Prescribed fire, however, can be successfully used in aspen stands. Fuels and flammability vary considerably and it is important to properly appraise fuels, flammability, and time of burning to help determine the proper conditions. Guides to help managers are available (Brown and Simmerman 1986).

Because of active fire suppression efforts, succession in many stands has progressed to the point where conifers are codominant. In stands where the coniferous component is significant, prescribed fire could be used in conjunction with timber harvesting. Consider, for example, commercially logging large conifers, felling and leaving smaller conifers, and burning slash accumulations to top-kill the aspen component of the stand. This scenario would approximate natural fires that resulted in successful aspen regeneration.[6]

The optimum size of individual areas to be burned depends on several factors including management objectives, size of the original stand, and other constraints that may be imposed by the prescription. Treated areas that are too small may result in concentrating browsing animals to the point where aspen regeneration is eliminated. This is a particularly vexing problem in the Southwest (Jones 1967).[6] Fencing treated areas until aspens grow out of reach of browsers would be expensive and useful only in specific, critical cases. Few other guidelines are available. Mueggler and Bartos (1977) stated that, without control of ungulate use, clearcutting or burning less than 12 acres (4.9 ha) might be futile. In the Southwest, treating a series of larger areas (15-25 acres [6-10 ha]) may be warranted, provided proper attention is given cover/forage patterns and that no single dimension exceeds 660 ft (201 m). In either case, several such areas should be treated in the same general area to distribute browsing pressure as much as possible.

[6]*Shepperd, W.D.; Crouch, G.L. 1988. Aspen management evaluation-7/25-29/88 observations and recommendations. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station (unpublished report).*

Management Implications

Prescribed fire offers an economically and environmentally acceptable way to revitalize decadent or late seral aspen stands. It can be used by itself or coordinated with some form of timber harvest. Fire, when properly applied, will increase forage production, diversity of the forage base, and diversity of cover types. Reviews have indicated a great reduction in the rate of fire in aspens in the West. Most stands are becoming decadent or dominated by conifers (DeByle et al 1987). If aspen forests are to be sustained, the use of prescribed fire is recommended. In fact, if the present aspen acreage in the Southwest is to be maintained or increased, manipulation of older stands is imperative (Patton and Jones 1977).

Riparian-Stream Ecosystems

Little literature exists on the specific effects of fire on riparian-stream ecosystems. Most of the focus has been on the hydrological and erosional responses to removal of vegetation by fire and the resultant effects on sedimentation and water quality (e.g., temperature, nutrients) (Anderson 1976, Tiedemann et al. 1978, Wright and Bailey 1982). Tiedemann et al. (1978), in reviewing the effects of fire on water, discussed flows, water quality, sedimentation, water temperature, nutrients, and "aquatic habitat response." Under the latter subject area the authors addressed aquatic macroinvertebrates, algae, periphyton, and phytoplankton. The other factors discussed in this review inherently were "aquatic habitat" factors and how these, in turn, affect fish populations. However, neither this work, nor Lyon et al. (1978) specifically addressed the effects of fire on fishes because no such studies were available. Lyon et al. (1978) suggest that increased soil erosion and input to streams may affect fishes directly by reducing spawning

success or indirectly by reducing their food supply.

Because no specific studies are yet available on fire effects on fishes, one has to approach the problem from a habitat response. That is, what are the influences of either wild or prescribed fire on near and instream habitat and, in turn, what are the probable effects of these influences on fishes based on past and current results of fish-habitat research? Wright and Bailey (1982) suggest fire increases streamside deciduous vegetation, thereby providing increased cover and food supply for fishes. Nutrient increases are seldom toxic and benefit primary and secondary production in streams.

In summary, Tiedemann et al. (1978) dismissed water quality changes and increased nutrient inputs to streams caused by fire as important factors affecting water resources. We agree generically; however, large fires have the greatest potential for causing damage to water and its inhabitants. A large, hot wildfire, as a result of convectional heating and input of burning debris and ash, could conceivably increase stream water temperatures to a lethal level for coldwater salmonid fishes. This elevation of water temperature would have to be present only for a short (less than an hour) period of time (Lee and Rinne 1980). Although the recent Mt. Saint Helens volcanic eruption was of a much greater magnitude than a wildfire, it does demonstrate, in principle, that fish kills could result in smaller, montane streams because of heat and ash.

In agreement with Tiedemann et al. (1978), we suggest that instream and streambank sedimentation and hydrological response, singularly or in combination, are two primary factors that may be altered by either wild or prescribed fire. The degree of the response depends upon the size and extent of the fire and the topography and soils of the area.

Almost three decades ago, Cordone and Kelley (1961) implicated the probable effect of inorganic sediment on aquatic organisms including fishes. Currently, the effects of fine sediment on fishes and their habitat is a proposed priority research emphasis for the USDA Forest Service. Much research has been directed at this problem in the last 25 to 30 years; however, most results have come from laboratory studies (Chapman 1988, Everest et al. 1987).

Rinne and Medina (1988) suggested the influence of fine sediment on trouts and, to a lesser extent, aquatic macroinvertebrates in several first order, montane streams in central Arizona. Based on the literature and recent research results, fine sediment, fishes, and aquatic macroinvertebrates are useful indicators of condition of riparian-stream habitat as influenced by land management activities (Rinne, in press). Prescribed fire certainly can be classified as a management tool; and, if extensive enough relative to watershed area, it could affect fishes and their habitats.

Management Implications

Because no information is available on fire effects on fishes, we recommend, as do Bock and Bock (these proceedings) for low-elevation riparian areas, that land managers not burn southwestern riparian habitats. Some information is available for riparian sites of the northern Rocky Mountains (Gordon 1976), and fire has been suggested as a tool to improve riparian and riparian-like communities in the northern Great Plains (Severson 1981). No information is available for the relatively more fragile riparian areas of the Southwest. Research is needed to delineate both the immediate direct and subsequent indirect effects of fire on fish and their habitats. However, the rarity and valuable nature of riparian areas does not provide the luxury of studies designed to evaluate fire effects on these areas. Alternatively, researchers and land managers must cooperate to identify and opportunistically study wildfire events that either directly or indirectly may affect riparian stream ecosystems.

Summary

Fire has historically occurred in the Great Basin Conifer Woodlands, Montane Conifer Forests (including the ponderosa pine and mixed conifer series), and the Rocky Mountain Subalpine Forest. Although fire frequencies varied among these biotic communities, fire was an active force in maintaining a dynamic and constantly changing series of seral communities. Presumably, the resultant diversity benefitted wildlife populations in the long term.

In Great Basin Conifer Woodlands, prescribed fire can be used to alter composition and increase production of understory vegetation, stimulate shrub growth, and open closed canopies. It is commonly difficult to burn pinyon-juniper stands with dense canopies because, with no understory, fires do not carry easily. Openings can be created in these stands by burning individual trees or small areas where an understory does exist.

The most prevalent use of fire in ponderosa pine stands has been for fuel reduction. Concurrently realized wildlife benefits have included short-term gains in nutritional contents of forages and increased forage production because of litter and duff removal and reductions in competitive small trees. Ancillary benefits include stimulation of shrub growth. To maximize wildlife benefits, 25% to 33% of the forest floor should be burned in patches annually or at a frequency of every 3 to 4 years. While prescribed crown fires, which create early seral stages, may be difficult to apply, similar benefits may be achieved by using fire following timber harvest.

Mixed conifer and spruce-fir stands were subject to natural fires

on a less frequent basis than were pinyon-juniper or ponderosa pine stands.

Fire, however, is necessary to recreate seral stages within these forests, especially aspen. Only 7% of the aspen stands in Arizona and New Mexico are in seedling or sapling stages; hence, rejuvenation of older stands is essential. It is difficult to burn aspen stands, but new guidelines are available to assist managers. Fire can also be used in conjunction with timber harvest to recreate aspen stands.

The use of prescribed fire in southwestern riparian or other streamside habitats is not recommended at this time. While research has not directly addressed the effects of fire on these habitats, peripheral evidence indicates most effects would be detrimental, both to riparian and aquatic ecosystems. Research is needed to assess effects of fire, not only in riparian habitats, but also to assess how burning other habitats in the watershed may influence riparian and aquatic systems.

The primary purpose of using prescribed fire for management of wildlife habitats is to create diversity. Habitat diversity resulting from properly planned fires may be expressed by changing composition of the forage base, by manipulating nutritive contents of forage plants (including shrubs), and by altering the overstory, thereby creating a series of seral communities. Objectives can be realized by careful planning, which may include using fires of variable intensity, alone, or in conjunction with timber, or fuelwood harvest, or in fuel reduction programs.

Although we have stressed managing for diversity, research has yet to define the optimum patterns or "habitat mosaics" that would best serve wildlife populations. Defining optimum pattern arrangements, spatially and in the context of time (seral stages and rates of succession), is an intriguing problem certainly worthy of our attention.

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Fire Effects on Ponderosa Pine Soils and Their Management Implications[1]

W. W. Covington and S. S. Sackett[2]

Abstract.—Fire in southwestern ponderosa pine induces changes in soil properties including decreasing the amount of nutrients stored in fuels (forest floor, woody litter, and understory vegetation), increasing the amount of nutrients on the soil surface (the "ashbed effect"), and increasing the inorganic nitrogen and moisture content in the mineral soil. Soil temperatures are increased above lethal levels under heavy fuels, probably killing both microbes and roots. Soil pH appears not to be substantially affected by burning in this forest type. The greatest changes in soil properties occur where heavy fuels are consumed; therefore, the magnitude of the impacts for most soil characteristics should decrease along the sequence from piled slash, to old growth substands, to pole substands, to sapling stands.

Fire has played an important role in southwestern ponderosa pine *(Pinus ponderosa)* forests from pre-settlement times. Fire exclusion was the general protection policy throughout the first half of this century. Now, prescribed burning is being used in southwestern ponderosa pine forests, in large part as a response to extensive crown fires resulting from heavy fuel accumulations during the earlier fire suppression period.

Wright (1978) and Lotan et al. (1981) provide general information on fire effects on vegetation in ponderosa pine forests. Both of these references will lead the reader to earlier works. The effects of fire on soils in ponderosa pine have not been synthesized before this. However, Wells et al. (1979) present a general review of fire effects on soils in North America and it has several references to fire effects in ponderosa pine. Many of the general principles drawn from fire effects literature in other forest types are discussed by Covington and DeBano (this volume). We will not repeat those discussions in this paper.

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[2]Professor, School of Forestry, Northern Arizona University, Falgstaff, AZ 86011; and Research Forester, Pacific Southwest Forest and Range Experiment Station, at the Station's Forest Fire Laboratory, 4955 Canyon Crest Drive, Riverside, CA 92507.

This paper synthesizes information regarding the effects of fire on soils in southwestern ponderosa pine. The intended audience for this paper is managers and others interested in ponderosa pine fire management in the Southwest. Where specific studies of fire effects on some soil properties in the Southwest are lacking, we have drawn inferences from studies conducted in other regions and other forest types. We have clearly identified where the inferences are based on research in other types, but the reader should be cautious about these extrapolations.

SOUTHWESTERN PONDEROSA PINE FORESTS

Ponderosa pine forests cover almost 11 million of the 26.5 million acres of commercial forest land in Arizona, Colorado, New Mexico, and Utah (Schubert 1974). Although individual ponderosa pine trees occur at elevations below 1,830 m (6,000 ft) and above 3,050 m (10,000 ft), ponderosa pine forests reach their best development in the Southwest between 2,130 and 2,380 m (7,000 and 7,800 ft) (Schubert 1974). At lower elevations ponderosa pine forests usually abut pinyon-juniper woodlands or chaparral; at higher elevations they grade into mixed conifer forests. Southwestern ponderosa pine forests are characteristically a mosaic of small, even-sized patches of: (1) large old-growth trees, which became established before European settlement in the 1870's, (2) intermediate sized trees, which became established around the turn of the century, and (3) small saplings, which became established around 1914-1930 (Cooper 1960, Schubert 1974, A. White 1985). The following tabulation presents general size and age classes for these substands as suggested by Covington and Sackett (1984).

Substand type	d.b.h. (inches)	Age (years)
Old growth	>11	200-500
Pole	4-10.9	80-120
Sapling	0-3.9	60-70

Ponderosa pine is the dominant tree species, but there is often an admixture of Gambel oak (*Quercus gambelli*), pinyon pine *(Pinus edulis)* and junipers (*Juniperus* spp.). The dominant understory species are usually grasses such as Arizona fescue *(Festuca arizonica)*, mountain muhly *(Muhlenbergia montana)*, and blue grama *(Bouteloua gracilis)*. Shrubs occurring in this type include buckbrush *(Ceanothus fendleriana)*, locust *(Robinia neomexicana)*, and currants (*Ribes* spp.).

Southwestern ponderosa pine forests occur on a wide variety of soil types derived from igneous, metamorphic, and sedimentary parent materials. Soil depths and textures

also vary greatly, with textures ranging from stony, cobbly sands and loams through clays and clay loams. Ponderosa pine site index is generally positively correlated with soil depth (Cox et al. 1960, Myers and Van Deusen 1960) and with the proportion of material less than 0.2 mm in diameter (Schubert 1974).

FIRE ECOLOGY

Frequent low intensity natural fires have shaped ecological patterns and processes in southwestern ponderosa pine ecosystems for millennia (Biswell et al. 1973, Cooper 1960, Pyne 1984, Weaver 1951). Natural fires burned through pre-settlement ponderosa pine forests in northern Arizona at intervals of 2-5 years (Dieterich 1980) However, following settlement the natural fire regime was disrupted. Beginning in mid-1870's, heavy livestock grazing decreased herbaceous fuel continuity and broke the continuity of the forest floor. Thus grazing, coupled with active fire suppression from the early 1900's, has resulted in the absence of natural fires from many of these ecosystems for over 100 years.

This fire exclusion has caused major changes in the fire regime and hence in the spatial pattern and ecosystem processes of these ecosystems (Kilgore 1981). Ponderosa pine forests have changed from open savannahs with relatively high herbaceous productivity to closed forests, where most of the production is in the trees. In addition to this shift in production, there is evidence for stagnation in both decomposition and nutrient recycling processes because of fire exclusion. The long absence of natural fire has been blamed for everything from dangerously high fuel loads (with an associated shift from frequent, low intensity understory fires to infrequent high intensity crown fires) to reduced productivity and stagnated nutrient cycles (e.g., Arnold 1950, Cooper 1960, Biswell 1972, Weaver 1974, and Covington and Sackett 1984).

Nitrogen volatilization from fuels consumed by burning has been a concern in ponderosa pine (e.g., Klemmedson 1976, Covington and Sackett 1984). On the other hand, fire exclusion in southwestern ponderosa pine may degrade the N economy of ponderosa pine forests by allowing litter to steadily accumulate, blocking the recycling of organically bound N into inorganic N forms (ammonium and nitrate), which are available for plant uptake (Biswell 1972, Covington and Sackett 1986). Since ponderosa pine ecosystem productivity is limited by low N availability (e.g., Wagle and Beasley 1968, Heidmann et al. 1979, Cochran 1979, Powers 1980), a major concern has been fire effects on inorganic N concentrations in the mineral soil.

Nitrogen distribution in ponderosa pine forests is correlated with the general pattern of the overstory vegetation. Overstory, litterfall, and forest floor biomass are greatest in old growth patches, intermediate in pole-sized patches, and least in sapling patches (Covington and Sackett 1986, Ryan and Covington 1986). Estimates of unburned forest floor characteristics of ponderosa pine substands at the Chimney Spring Interval Burning Study Area (Covington and Sackett, unpublished) are presented in the following tabulation.

Characteristic	Old growth	Pole	Sapling
Forest floor (g/m^2)	106,900	55,200	35,200
Litterfall (total <10cm diameter, in g/m^2)	4,500	3,090	2,030
Decomposition rate (k)	0.042	0.056	0.058

Understory vegetation biomass is lowest immediately under the canopy of the old growth trees, highest in the openings between patches, and intermediate in the pole and sapling patches (Harris and Covington 1983, Oswald and Covington 1984, Andariese and Covington 1986). Thus, old growth patches have the greatest amount of nitrogen susceptible to burning, pole patches have an intermediate amount, and sapling patches have the least.

These differences are slight in unburned stands (Covington and Sackett 1986).

Substand type	Ammonium	Nitrate
	(ppm)	
Old growth	1.51	0.18
Pole	0.96	0.01
Sapling	0.64	0.01

Nitrogen concentrations in the mineral soil are highest in old growth patches, intermediate in pole patches, and least in sapling patches; within a patch, the greatest concentrations are in the top horizons, decreasing with depth (Ryan and Covington 1986, Covington and Sackett 1986). Distribution patterns for other nutrients are not available.

Fire Effects on Plant and Litter Nutrients

The immediate impact of burning on soil nutrients is conversion of much of the organically bound nutrients in the forest floor, woody debris, and herbaceous vegetation into their inorganic forms. Whether these inorganic nutrients remain on site as solids or are lost through volatilization depends upon the temperatures reached during the burning and the differential volatilization temperatures of the nutrients (Wells et al. 1979, DeBano, this volume). Nutrients with a relatively low volatilization temperature (N, P, and S) are likely to have some loss to the atmosphere in most fires, whereas nutrients with high volatilization temperatures (Ca, Mg, and K) are, for the most part, left behind in ash.

The greatest loss of volatile nutrients and the greatest ash deposits from the less volatile nutrients occur where fuels are highest, such as in the old growth patches or in piled

slash. (Covington and Sackett 1984, 1986; Ryan and Covington 1986). Klemmedson (1976) estimated the impacts of slash burning on N budgets in ponderosa pine near Flagstaff, Arizona. He found that approximately 58 kg/ha of N was lost from the slash and forest floor 10 months after burning. However, he pointed out that it was not possible from his study to determine how much of this unaccounted for N had actually been lost. Some nutrients volatilized from the forest floor and slash are transferred to the mineral soil (see below). To minimize risk of N loss, slash piles should be put where forest floor loads are low (Klemmedson 1976).

Fire Effects on Soil Nutrients

Soil Nitrogen

Inorganic nitrogen concentrations in the mineral soil have been shown to increase after burning. Burning increased soil ammonium by as much as twentyfold in ponderosa pine near Flagstaff (table 1).

Numerous processes may be involved in long term changes in N in the mineral soil, such as higher microbial mineralization after burning, leaching from ash, and decreased uptake because of root mortality, (e.g., Ryan and Covington 1986, Covington and Sackett 1986). However, measurements of soil N after burning in ponderosa pine indicate ammonium increases within the first 24 hours (Covington and Sackett, 1990). Transfer of ammonia from the burning fuel is the most likely source. Ammonium can be produced during a fire through pyrolysis of organic nitrogen compounds (Christensen 1973, DeBano et al. 1979, Mroz et al. 1980). Since N is transferred from the burning fuel to the mineral soil, unless increases in mineral soil N after burning are accounted for, one would tend to overestimate the amount of N lost by volatilization to the atmosphere.

Inorganic N increases were greatest in old-growth substands, intermediate in the pole substands, and least in the sapling substands (Covington and Sackett 1986; Ryan and Covington 1986). Fuel loads and proportion of forest floor burned were highest in the old growth, intermediate in the pole, and least in the sapling substands, and ammonium concentrations followed a similar pattern. This correlation between fuel consumption and ammonium concentration suggests that differences in post-burn ammonium concentrations were due to differences in fuel consumption (Covington and Sackett, 1990).

These increases in inorganic N concentrations in the mineral soil are short lived, in the absence of repeated burning. By 4 years after an initial burn in ponderosa pine near Flagstaff, AZ, differences between burned and control plots were slight (around 1 ppm for ammonium-nitrogen. (Covington and Sackett 1986). However, burning at shorter intervals (1-2 years) maintained inorganic N concentrations (ppm) in the mineral soil which were 6-15 times higher than controls (Covington and Sackett 1986).

Substand type	Controls	Burns: 2-yr interval	Burns: 4-yr interval
Old growth	1.51	10.64	2.83
Pole	0.96	11.37	2.02
Sapling	0.64	10.90	2.03

Table 1.—Inorganic nitrogen concentrations (ppm) in the 0-5 cm depth of the mineral soil immediately after burning (Covington and Sackett, 1990).

Substand type	Ammonium: Control	Ammonium: Burn	Nitrate: Control	Nitrate: Burn
Old growth	2.33	45.10	0.11	0.18
Pole	1.34	26.72	0.01	0.06
Sapling	1.29	8.28	0.01	0.02

Soil pH

Burning in some forest types may cause increases in soil pH (e.g., Grier 1975, DeByle and Packer 1976). Because most southwestern ponderosa pine forest soils are near neutral already, however, one would expect little change in pH after burning in this type. After prescribed burning in ponderosa pine near Flagstaff, Arizona, Ryan and Covington (1986) found no significant change in soil pH on a basalt site (pH ranged between 6.2 and 6.5). Campbell et al. (1977) also found no difference in pH for soils with sedimentary parent material between burned and unburned soils after a wildfire in ponderosa pine in northern Arizona (pH was 5.9). However, Fuller et al. (1955) found an increase of up to 1.1 pH units in both duff and mineral soil following burning in ponderosa pine on a sedimentary derived soil. It is not clear why Fuller et al. (1955) found changes while Campbell et al. (1977) and Ryan and Covington (1986) did not.

Soil Moisture

By changing the amount and type of vegetation and the forest floor, as well as the soil texture and wettability, fire can cause both short and long term changes in soil moisture relations. Fire in southwestern ponderosa pine has been shown to increase soil moisture content (Milne 1979, Ryan and Covington 1986, Ower 1985, Haase 1986). Campbell et al. (1977) found that runoff efficiency

increased from around 0.8 percent for unburned areas to 3.6 percent from areas severely burned by a wildfire in ponderosa pine. Peak flows were almost 400 times greater from the severely burned areas. They attributed these effects to reductions in vegetation and forest floor cover and to the formation of hydrophobic layers in the soil; infiltration was only 2.5 cm/hr on severely burned areas as compared with 6.9 cm/hr on unburned areas.

Soil Temperature

Fuel load, fuel moisture, amount of fuel consumption, soil moisture content, and soil texture all influence the amount of total heating and the peak temperatures reached in the mineral soil after burning (Wells et al. 1979). The amount of fuel consumed is the single most important factor in determining temperature change with burning. Under slash, natural woody debris, or the deep forest floors under old growth patches, soil temperatures can be quite high after burning. Sackett (unpublished data) has found lethal temperatures to a depth of 15 cm where heavy forest floors were almost completely consumed.

Over the longer term, burning can alter soil temperatures indirectly because of fire-induced changes in the albedo and shading of the soil. Darker, unshaded soils with little forest floor would tend to have warmer soils with wider swings in diurnal temperature than unburned soils (Wells et al. 1979). At a site near Flagstaff, Milne (1979) observed that soil temperatures were higher by 4-5 degrees centigrade at 6-15 cm depth on burned plots 1 year after burning compared with control plots.

Soil Microbes

Burning can alter microbial activity either directly through sterilization or indirectly through changes in soil temperature, moisture, and pH, or by changing organic matter quality and allelopathic properties. Little direct evidence of the effects of fire on soil microbes in ponderosa pine is available. However, forest floor decomposition was increased after prescribed burning in ponderosa pine near Flagstaff (Covington and Sackett 1984).

There is also evidence in the literature for increased microbial nitrification after prescribed burning in ponderosa pine. Using a laboratory incubation method, C. White (1985) found substantial increases in nitrification 6 months after prescribed burning in ponderosa pine in New Mexico. Higher nitrification *in situ* might be caused by warmer soil temperatures and higher soil moisture conditions after burning (Milne 1979, Ryan and Covington 1986), as well as by fire-caused decreases in allelopathy. Lodhi and Killingbeck (1980) found evidence for allelopathic inhibition of nitrification in both the forest floor and the mineral soil of ponderosa pine stands. C. White's (1985) results provide evidence for the role of fire in denaturing these allelopathic agents.

The high temperatures reached where slash piles, woody debris, and heavy forest floors are consumed undoubtedly cause extensive mortality of some microbes. The delay in nitrate increases after burning in ponderosa pine may be attributed to sterilization effects on nitrifiers (Ryan and Covington 1986, Covington and Sackett, 1990). As noted by Dunn et al. (1979), sensitivity to temperature is greatest for fungi, intermediate for nitrifiers, and least for heterotrophic bacteria.

IMPLICATIONS FOR PRODUCTIVITY

An important question that needs to be answered to assess the impacts of burning on ecosystem processes is this: What is the fate of the additional soil N availability? Some of the nutrients are undoubtedly leached from the site. However, since most of the inorganic N is in the ammonium form, which is tightly held on cation exchange sites in the soil (Ryan and Covington 1986, Covington and Sackett 1986), one would expect little to be leached from the ecosystem. Since these soils rarely have anaerobic conditions, denitrification losses would also be minimal. The most likely fate of the additional soil N availability is uptake by vegetation and immobilization by microbes. Evidence for uptake by vegetation comes from observations of higher N concentrations (mg/g) in grasses, pine needles, and needle fall on burned sites (data are from Harris and Covington 1983 and Covington and Sackett, unpublished).

Component	Nitrogen	
	Control	Burn
Grasses	13.9	18.9
Pine needles	11.7	13.1
Needle fall	3.4	4.3

The large increases in available N in the mineral soil after burning are likely the primary cause of (1) increased seedling establishment of both herbaceous vegetation (Vose 1984, Vose and White 1987) and ponderosa pine (Sackett 1984), and (2) higher herbaceous production and greater foliar N concentrations (Harris and Covington 1983, Oswald and Covington 1984, and Andariese and Covington 1986). This mechanism is supported in part by results from Harris and Covington (1983) who found the greatest increases in foliar N in their old growth substands with moderate increases in poles and saplings, consistent with the pattern for soil N described above.

Increased nitrogen availability after burning may enhance revegetation after burning in ponderosa pine ecosystems. The most severe fires and the greatest understory (Vose 1984, Vose and White 1987) and overstory (personal observation) mortality typically occur in the old

growth patches where fuels are the heaviest. In these same patches, available N is the highest after burning. This nutrient enrichment enhances revegetation and recovery of the severely burned sites.

Revegetation of the severely burned old growth substands is rapid (Vose 1984, Vose and White 1987). Growth rates of individual plants can be exceptional. Sackett (1984) reported 6-yr old ponderosa pine seedling heights of up to 60 cm on nearby prescribed burned plots in contrast with a normal height of 15-20 cm for 6-yr old ponderosa pine seedlings on similar unburned sites.

These diverse lines of evidence point to the conclusion that prescribed burning in southwestern ponderosa pine increases N availability and ecosystem productivity.

SUMMARY AND MANAGEMENT IMPLICATIONS

Burning in southwestern ponderosa pine increases soil nitrogen and moisture availability. These increases in nitrogen result in higher nutrient concentrations both in the understory and in the overstory vegetation. Greater ponderosa pine seedling establishment and growth as well as increases in understory production may be attributed in part to this increased N and moisture availability.

In the absence of repeated burning, these ameliorated soil conditions appear to be short lived. At one site, burning at a 2 year interval (which approximates the natural fire regime for that site) seemed to maximize soil N availability.

The effects of burning on forest floor mass, soil N, and understory production vary widely among old-growth, pole, and sapling patches. Burning prescriptions for ponderosa pine that fail to account for this spatial variability in both pre-treatment conditions and post-treatment response ignore a fundamental ecological characteristic of this type and are likely to produce unforeseen consequences. This is especially important in decisions regarding reintroducing natural fires, managing wilderness, or maintaining old growth.

For stands which, because of harvesting or wildfire, do not have this spatial heterogeneity, the land manager can now better predict the consequences of prescribed burning. However, caution should be exercised in extending the results from the literature, especially for the old growth and sapling substands (which cover small areas), to larger areas (e.g., hundreds of hectares).

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Playing with Fire: Effects of Fire in Management of Southwestern Recreation Resources[1]

Jonathan G. Taylor[2]

Abstract.—Recreation is of increasing importance in forest environments. Fire has both short-term effects, trail closures, smoke impacts; and long-term effects, residual "scars," potential hazards, on forest recreation. The general public is gaining sophistication in understanding forest fires. Fire managers must educate themselves concerning public response and fire knowledge.

The Importance of Recreation in Forest Environments

Outdoor recreation is continuing to increase in the United States. Hendee et al. (1977) estimated that public use of wilderness areas will increase, in the 40-year period from 1960 to 2000, by nearly tenfold. A report by the Heritage Conservation and Recreation Service (HCRS; 1979) showed rapid growth in such outdoor recreation activities as skiing, snowmobiling, canoeing, and off-road-vehicle use as well as tremendous growth potential in the area of primitive camping. The President's Commission on Americans Outdoors (1987) reported persistent vigorous interest in a broad spectrum of outdoor recreation; for example, there were increases from 1960 to 1982 in canoeing (515%), bicycling (382%), camping (240%), hiking and backpacking (199%), and walking for pleasure (132%). Of the federal land-management agencies, the U.S. Forest Service continues to provide more total visitor time than any other federal agency—approximately 2.5 billion visitor hours in 1984.

The Commission Report also showed that outdoor recreation trips are tending to become shorter in both time and distance. Short trips are rapidly supplanting older patterns of 2-3 week, long-distance vacations (fig. 1). The American public is traveling shorter distances to recreate more often.

The President's Commission also convened expert panels to identify major societal trends that might have important effects on the future of outdoor recreation. In this context, the following 10 trends were believed to be the most significant:

1. **Changing social and demographic composition**—American society is aging.

2. **Fluctuating energy availability and cost**—These create uncertainties for recreation and tourism.

3. **Technological innovations**—These change the demands on type and location of outdoor recreation (e.g., wind surfing, mountain biking).

4. **Shifts in political power closer to the people**—The Federal government is releasing control and financing, increasing authority at local levels.

5. **Increased accountability of institutions and leaders**—This parallels the "megatrend" of increasing participatory democracy.

6. **Concern for the environment**—A continued high emphasis on environmental protection, with specified emphasis toward environmental health and safety, is evident.

7. **Creation of innovative partnerships**—Public and private cooperative efforts in outdoor recreation are increasing.

8. **Shifts in economic strengths and weaknesses**—Employment shifts from manufacture to service and information industry affect time and money available for outdoor recreation.

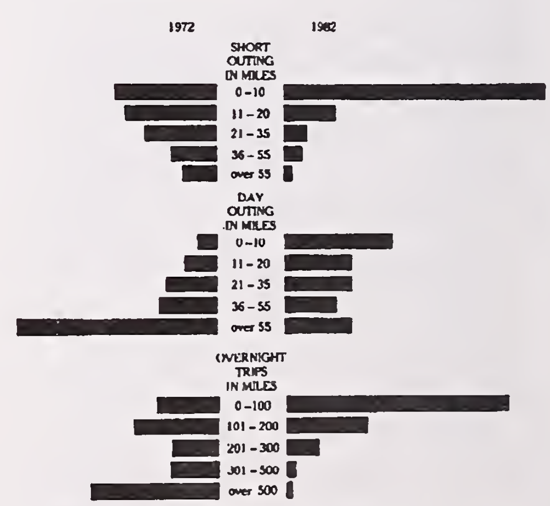


Figure 1.—Recreation trip distance by type of outing, 1972-1982. (President's Commission on Americans Outdoors 1987).

[1]*Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Research Social Scientist, National Ecology Research Center, U.S. Fish and Wildlife Service, Fort Collins, Colo.*

9. **Changes in recreation and leisure**—Technology, fashion, and fad, are changing.

10. **Changes in transportation systems**—This influences how and where Americans travel for pleasure.

These recreational and societal trends have implications for the management of forest recreation resources. Western forest managers should anticipate, at least for the near term, increasing proportions of local and regional people using their recreational resources. Further, because people are taking shorter recreational trips, overall recreational use may be spread out over a broader prime season, with less concentration on the traditional overuse weekends such as Memorial Day, July 4th, and Labor Day.

Turning specifically to the Southwest, how important is recreation in forest areas to the public in this region? In a 1981 telephone survey of 1,200 residents of Tucson, Arizona, Zwolinski et al. (1983, see also Cortner et al. 1984) asked respondents to rate the relative importance of forest areas in comparison with other resources. More than 87% responded that they considered forest areas as "very important" and 97% rated forest areas as somewhat to very important. Fifty-three percent reported that they "regularly participate in forest activities" including hiking (50%), camping (47%), picnicking (28%), fishing (19%), hunting (14%), backpacking (6%), and skiing (5%), all of which are forest recreation activities. Three percent reported use of forest areas because they had a forest home, and 3% reported that they worked in the forest.

Respondents were asked the importance "to you personally" of a series of resources found in the forest. Nearly half (47%) of the survey respondents rated "opportunities for outdoor recreation, such as picnicking, camping, and hiking" as "extremely important" on a scale of 1 = not at all important to 10 = extremely important; 76% rated recreation as 8, 9, or 10 on this 1-to-10 scale. Respondents rated recreation as third in forest resource importance, after "food for wildlife," (87%, 8-10) and "protection of water supplies" (84%, 8-10). Recreation was rated more important, by this general public sample in the Southwest, than "lumber and other wood products" (66%, 8-10), "food for livestock" (47%, 8-10), "firewood" (33%, 8-10), and "opportunities for hunting" (29%, 8-10).

The importance attributed to outdoor recreation in forest environments by both the national and regional publics suggests that our National Forests cannot be managed strictly on a cost-effective basis for production of such payback commodities as timber and livestock. To place higher priority on timber, livestock, firewood, and hunting production than on wildlife forage, watershed protection, or recreation would be to directly contradict public priorities. Because of these changing and interacting priorities, the U.S. Congress passed the Multiple Use Sustained Yield Act in 1960 and the National Forest Management Act in 1976. As will be seen later, however, fire managers often follow older, traditional resource value priorities in making fire-risk decisions.

Short-Term and Long-Term Effects of Fire Management

Fire can affect recreation in many ways; these can be separated into two categories: immediate, short-term effects and long-term effects. Short-term effects include the direct impingement that fire can and will have on people's recreation decisions, such as "East Rim Trail CLOSED due to Fire." A second type of short-term effect is the impact of smoke from forest fires on outdoor recreation in the vicinity.

Long-term effects include the residual impacts, or "scars," left by fire and the ways these affect scenic quality or the recreational acceptability of a forest area. Other long-term effects of fire on recreation include potential tree-fall, "quick soils," and loosened rocks that may be long-term safety hazards to recreationists. In this context, it is important not to make the mistake, all too common, of assuming that recreational acceptability and scenic quality are one and the same. They are not equivalent; indeed, acceptability of long-term fire effects differ from one form of outdoor recreation to another.

Immediate, Direct Effects of Fire

Resource management agencies are shifting their approaches from all-out suppression to fire management, including prescribed burning with attention to the ecological roles played by fire in different forest types. These agencies are concerned whether the public, indoctrinated with Smokey the Bear ethics, will be able to accept these changes (Smith 1986). Omi and Laven (1982) reported that public knowledge and acceptance of prescribed fire in recreational wildlands lagged behind implementation for several reasons: interpretations of the Smokey the Bear message that all fires are detrimental, public concern and legislation related to air quality, and the lack of consensus among forestry professionals about the appropriate timing and specific uses of fire. The direct effects on the general public of different fire management practices should be expected to vary greatly depending on the level of public knowledge and acceptance of these practices.

Conservation organizations, the news media, and the informed public are learning more about the natural role fire plays in forest environments. The Rocky Mountain News (August 29, 1988) covered the extensive fires in Yellowstone National Park and included an informative article on

the beneficial role fire plays in these forests: "Forest fire and the rebirth of the ecosystem" (p. 4). National Public Radio's "Morning Edition" broadcast a fairly sophisticated discussion of the ecological role of fire in Yellowstone National Park (Sept. 12, 1988). Surveys also showed growing public sophistication in understanding the natural role of fire and in accepting new fire management practices—perhaps more than many professional foresters would believe.

Public Knowledge of Fire Behavior and Effects

McCool and Stankey (1986) found that public knowledge of fire is increasing. Correct answers to fire knowledge questions increased from 53% to 64% between 1971 and 1984. Persistently, the lowest correct response rates were to questions about animal mortality and fire size. In the Southwest, both Taylor and Daniel (1982, 1984) and Zwolinski et al. (1983) tested public knowledge of fire behavior and effects. The resulting information from these surveys gives some indications of where the public is informed, where misinformed, and where amenable to education.

Taylor and Daniel (1982) developed and tested information brochures designed to educate the public on the effects of fire in ponderosa pine forests. Their results were mixed; support for prescribed burning increased and some specific areas of fire knowledge also increased, whereas other areas of knowledge did not. Baas et al. (1985) found that information about the use of fire, given to visitors to the Grand Canyon National Park, did not significantly increase their support for prescribed burning or their knowledge of fire effects.

Haug Associates, Inc. (1968), found that the adult American public believed that forest fires caused more damage than floods, hurricanes, or earthquakes. Concern centered especially around destruction of timber, wildlife, and homes and property. Consistently, the public seems to be most misinformed concerning the effects of fire on animal mortality. This item was among the questions with the fewest correct answers in the surveys by McCool and Stankey (1986), Stankey (1976), Taylor and Daniel (1982), and Zwolinski et al. (1983). Two dimensions to this misperception emerge from the southwestern research.

First, the Tucson public exhibited an unwarranted level of confidence in their misinformation about animals being killed by forest fires. Although only 8% of the Zwolinski et al. (1983) sample agreed with the "expert" group that "Few animals are killed by most forest fires," only 4% answered that they "did not know" how many animals typically were killed. However, in a paired-comparison test of the effects of their fire information brochures on fire beliefs, Taylor and Daniel (1982) found this dimension significantly changed by the information treatments (fig 2). The public is fairly confident in its misperception of how fire affects wildlife, but is capable of being swayed from that position through education. Smokey and Bambi have certainly overdone their jobs; but, perhaps because this is such a deeply felt issue, the public is quite educable on this dimension when new information contradicts strongly held beliefs.

Second, the public is misinformed about fire size and intensity. McCool and Stankey (1986) found this, along with animal mortality, to be consistently misperceived over their 14-year test period. Fully three-quarters of Zwolinski et al.'s (1983) sample believed that most forest fires are "very hot with tall flames," burning 100s or 1,000s of acres; however, 10% answered that they did not know average fire size or intensity. Only about 15% agreed with the expert group that fires ordinarily burn at moderate intensity and cover a few acres or less. These two parameters also could be changed through education in Taylor and Daniel's (1982) study, both shifting from the uninformed, more-severe assumption toward the expert, less severe position.

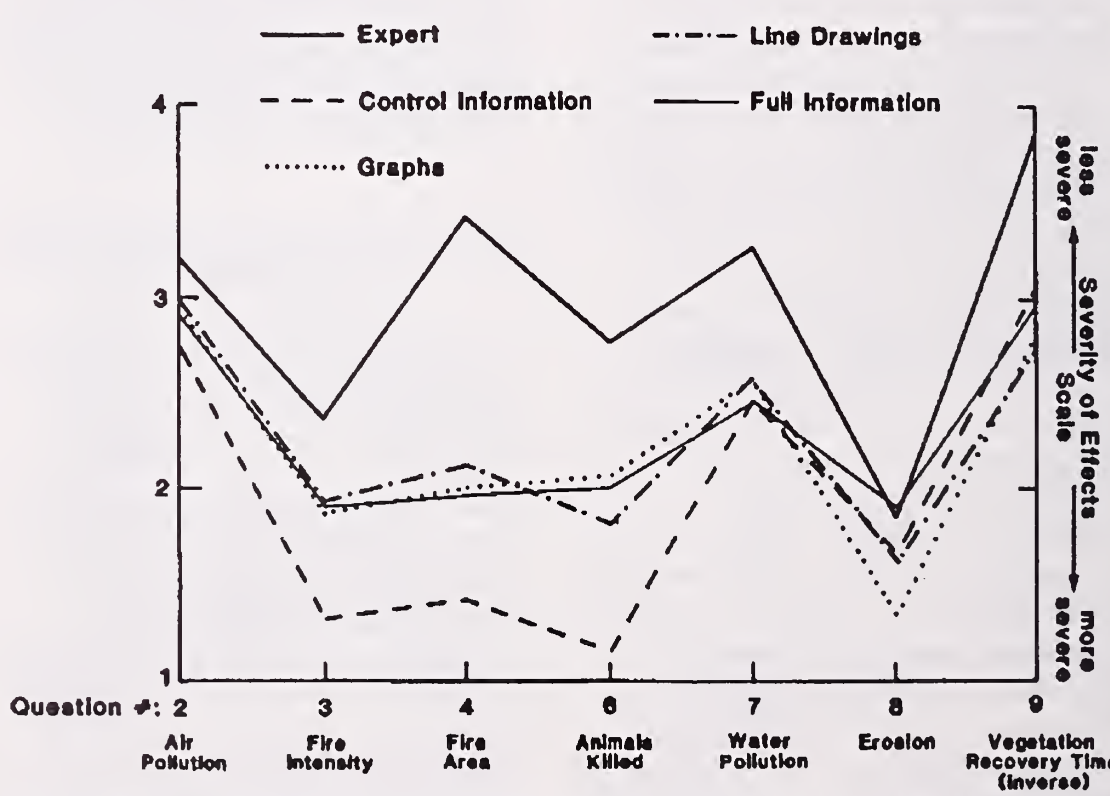


Figure 2.—Effects of information on public knowledge of selected fire items (Taylor and Daniel 1982).

A third area of misperception common to all four surveys drawn upon here (McCool and Stankey 1986, Stankey 1976, Taylor and Daniel 1982, and Zwolinski et al. 1983) is that most forest fires in the northern and southwestern United States are started through human carelessness. In both of these survey regions, lightning is the principal origin of forest fires. However, fewer than one-fourth (23%) of the people surveyed correctly identified this fire origin (Zwolinski et al. 1983), whereas more than two-thirds (68%) identified human carelessness as the principal origin. This parameter was not shifted for those respondents who read fire information brochures in the Taylor and Daniel study.

Survey results were mixed concerning public knowledge about the effects of fire on air pollution, water pollution, and erosion. Nearly half (45%) of the Zwolinski et al. (1983) sample agreed with the expert group that forest fires contribute only a minor portion of air pollution; the same agreement occurred in the Taylor and Daniel (1982) study. The Taylor and Daniel respondents also agreed with the expert group that severe fires result in moderate soil erosion; but the Zwolinski et al. sample tended to overestimate this effect, 53% believed that major amounts of soil erosion generally follow.

Contribution of forest fires to water pollution had a wider spread of beliefs; one-third (33%) agreed with the experts in Zwolinski et al. that minor water pollution resulted, but more than one-fourth (27%) believed that moderate water pollution resulted, and another one-third were evenly split between "No water pollution" and "Major water pollution."

The public seems to be fairly well informed that periodic light-intensity fire reduces the danger of subsequent severe fires. This was one of the items of important knowledge increase reported by McCool and Stankey (1986), whereas 65% agreed with the statement in the Zwolinski et al. (1983) survey. In Taylor and Daniel's (1982) survey, effects from information were not found for this item, because more than 90% of those not provided with fire information agreed with the statement to begin with.

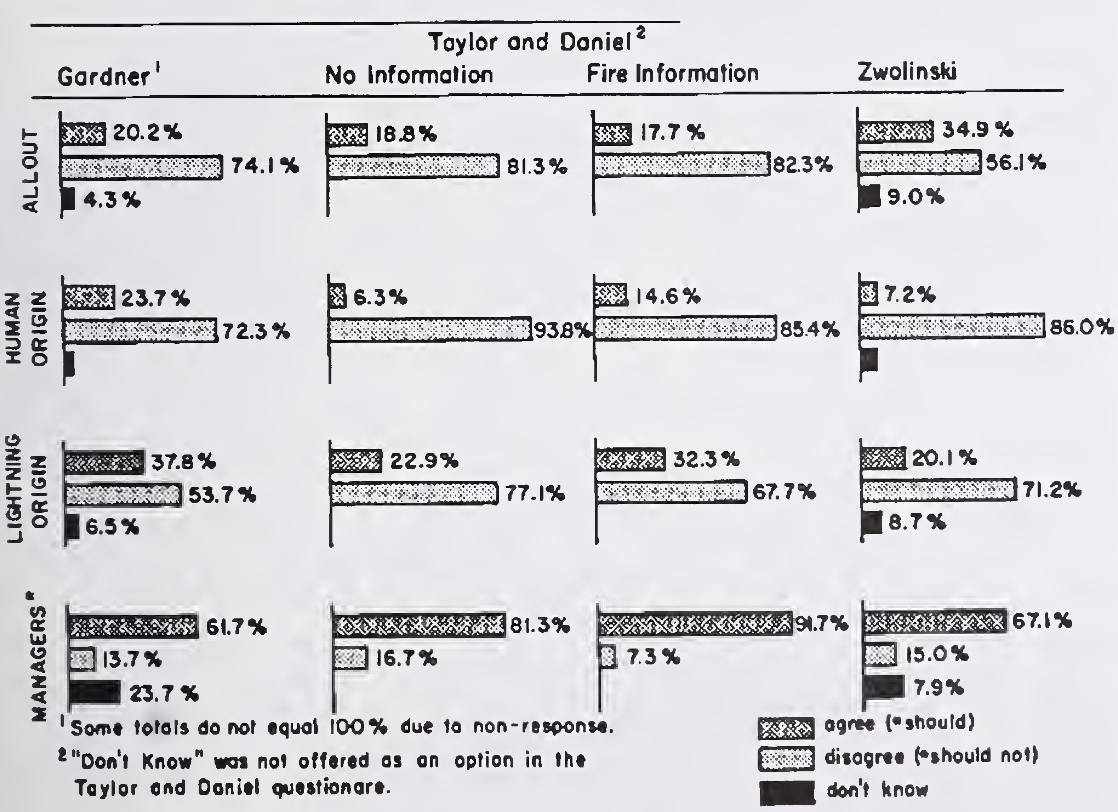


Figure 3.—Public support for complete suppression, allowing human-origininated or lightning fires to burn, and for managers burning out underbrush and debris.

Public knowledge of the effects of fire on forest ecosystems was mixed. McCool and Stankey (1986) reported that people's knowledge had increased concerning fires opening up meadows and grassy areas and that fire suppression could change the composition of plant species and reduce certain habitat. However, understanding that fire and fire suppression could effect changes in the ecosystem in general changed little over the years, and the effects of suppression on elk habitat remained one of the areas of lowest knowledge.

Taylor and Daniel (1982) and Zwolinski et al. (1983) asked their respondents whether an underbrush and debris fire would allow other tree species to replace pines or would tend to keep the area as a pine forest. More than one-third (35%) of Zwolinski et al.'s respondents answered "keep it pine," but 27% answered "replace pines," and 38% didn't know. This is one clear area of response to the information brochures in Taylor and Daniel's survey: slightly more than half of the uninformed group correctly identified "keep it pine," but 90% of each of the fire information treatment groups responded correctly. This is an aspect of fire that the public does not completely understand but has an interest in learning more.

Public Acceptance of Fire Management Practices

From Stankey (1976) and McCool and Stankey (1986), we know that the attitude of wilderness users toward fire management policies varies according to their level of knowledge about fire. These researchers demonstrated a significant increase by the public, over time, in tolerance of fire in wilderness. Most visitors (56%) to the Selway-Bitterroot Wilderness in Idaho in the initial survey in 1971 favored fire suppression. Most in the follow-up survey in 1984 (73%) favored the use of fire as a wilderness

management tool. This is a marked departure from the position reported by Hall (1972) that the strongly dominant attitude of North Americans was that fires were categorically bad.

Analysis of survey results from Gardner et al. (1985), Taylor and Daniel (1982), and Zwolinski et al. (1983) showed that the public is not only accepting of the use of prescribed fire, but is discriminating among different kinds of fire. When asked if all fires in the forest should be vigorously suppressed, two-thirds (67.3%) of these sample audiences, overall, disagreed (fig. 3). However, 78.5% believed that fires started through human carelessness should be suppressed, and 61.5% believed that fires started by lightning should be put out. Conversely, 82.2%, overall, believed that forest managers should periodically burn out underbrush and debris.

This suggests a degree of sophistication in public response to forest fire previously unknown to forest managers. Earlier studies (Hall 1972, Hendee et al. 1968) supported the idea that the general public viewed all forest fires as bad and supported vigorous suppression policies. But fire management is changing and so is public acceptance of fire policies. Carpenter et al.'s analysis (1986) of the "fire acceptance" data from the three surveys cited above demonstrated that the attitudes of the public shift toward ever-greater acceptance of fire in a forest environment as the nature of that fire is more completely explained and the degree of oversight is increased (table 1).

Carpenter et al. (1986) also conducted a log-linear analysis of data from these three surveys data to determine what sociodemographic or fire-knowledge factors might be significant in explaining differences in attitudes toward these different fire management situations. Four knowledge or belief factors emerged as significant in explaining attitudes toward fire management (table 2):

Table 1.—Attitudes (%) toward fire management approaches as they are more completely defined.

Management practice	Gardner	Taylor & Daniel	Zwolinski et al.	Total
Fires already burning should be allowed to burn if watched.				
Agree	—	20.8	22.2	22.0
Disagree	—	79.2	77.8	78.0
Fires burning underbrush and debris, but not tall trees, should be allowed to burn if watched.				
Agree	61.4	72.2	64.4	63.2
Disagree	38.6	27.8	35.6	36.8
Forest managers should periodically burn underbrush and debris in pine forests.				
Should	81.8	89.4	81.7	82.2
Should Not	18.2	10.6	18.3	17.8

Table 2.—Knowledge levels on significant explanatory variables: Beliefs about fire.

	Willing to accept fires that are[1]				
Beliefs	Already burning	Underbrush & debris	Set by managers	Prescribed burns	Knowledge[2]
Beneficial effects	++	++	++	++	Increased re: Nutrients Insects and disease Not changed
Origin of fire	++	++	++	++	Significantly, Low Not changed
Fire Area	++	++			Significantly, Low Not changed
Animal mortality	++	++			Significantly, Low

[1]Source: Carpenter et al. 1986.
[2]Source: McCool and Stankey 1986, Taylor et al. 1986.
++Significant explanatory variable

knowledge of beneficial effects of fire, of the most-common fire origin, of the size of most fires, and of animal mortality resulting from most fires.

The Relative Importance of Smoke

The second area of concern regarding the direct effects of fire on recreation is the influence of smoke generated by fires. However, this seems to be a somewhat elusive parameter in terms of public response and the effects on recreation resources. In the telephone survey of Tucson residents cited earlier (Zwolinski et al. 1983), respondents were asked the question, "If you were to see or know that there was a forest fire, how concerned would you personally be about each of the following?" (on a 1-to-10 scale where 1 is not at all concerned and 10 is extremely concerned). Figure 4 displays the responses to this question. Three distinct groupings of "extreme concern" to the public were noted. Losses of trees, wildlife, and food for wildlife all were rated "10" by more than half of the 1,200 respondents. Loss of recreation area, runoff and erosion, and loss of livestock food were of extreme concern to 30-40% of these Tucson residents. Fewer than 20%, however, consider smoke pollution from forest fire to be of extreme concern; fewer than 10% rated this item as 8 or 9 on the 1-to-10 scale.

In summer 1988, Habeck (1988) conducted a preliminary test of a public response survey concerning smoke from prescribed burning in the Bitterroot Valley of Montana. Although these results were preliminary and the sample quite small, it was proportionally drawn from four different communities in the valley and the results tabulated. Seventy-one percent reported that they did not perceive an air pollution problem from smoke, and 68% believed that prescribed burning was only a "minor" or "no factor" in valley air pollution. Sixty-four of the 69 respondents (92%) were willing to tolerate minor-to-moderate amounts of smoke as a result of prescribed burning.

Studies of economic valuation of air quality in several National Parks in the Southwest (Blank et al., 1978) showed that people consider maintaining or improving air quality in and around National Parks quite important. Local residents and park visitors were willing to pay $50 to $85 per household per year to achieve significant improvements in air quality. These studies also showed that people were willing to pay more for maintaining or improving air quality in pristine areas than in natural areas where some environmental degradation had already occurred. Crocker (1986) found that users of the Central Oregon Cascades Wilderness would be willing to pay an additional $2.00 per day of use to assure high visual quality.

These studies suggest that air quality, in outdoor recreation areas, is of serious concern to the American public, but that smoke from prescribed burning is not perceived to be a major factor in air quality dete-

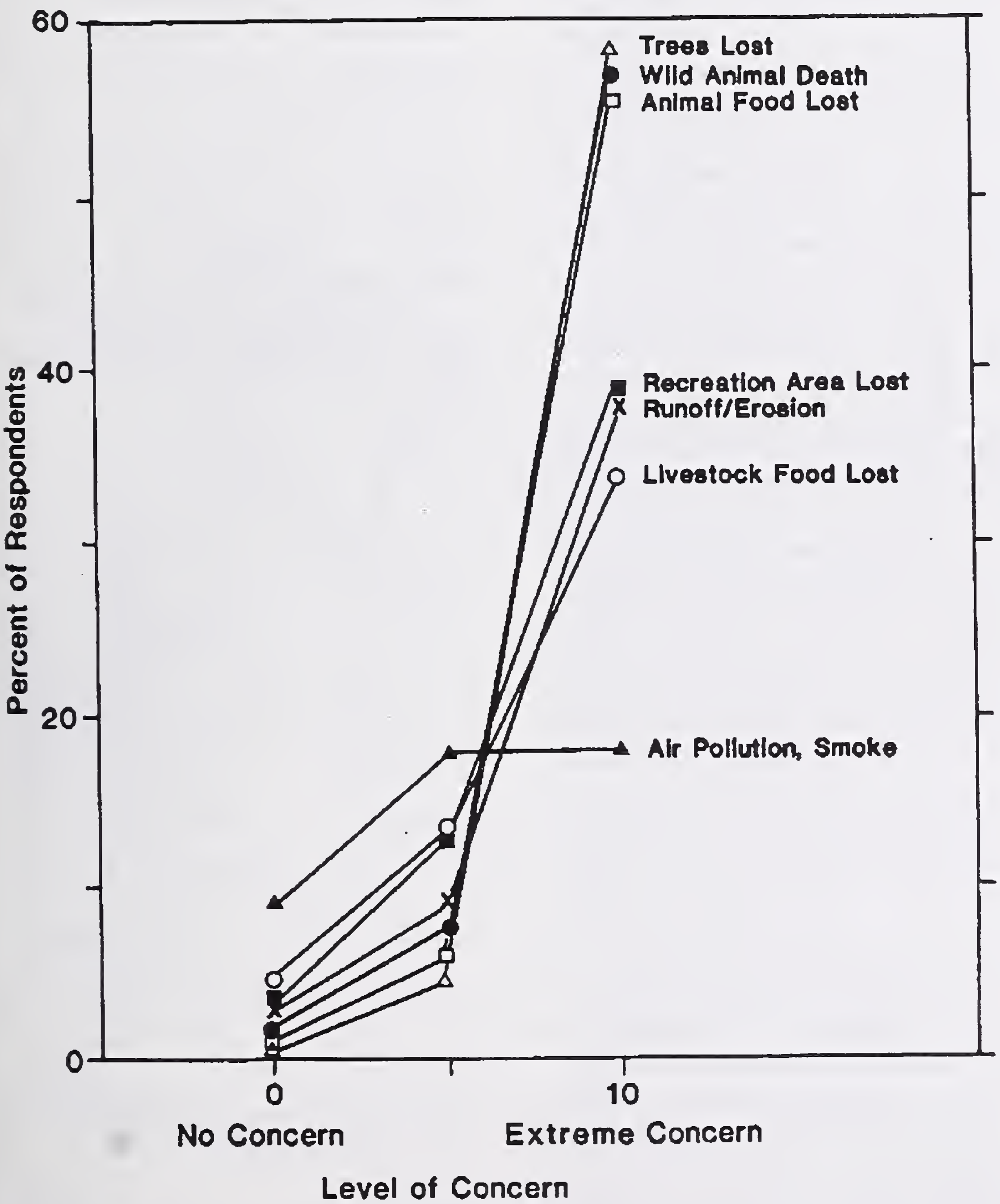


Figure 4.—Public concern for fire effects (Zwolinski et al. 1983).

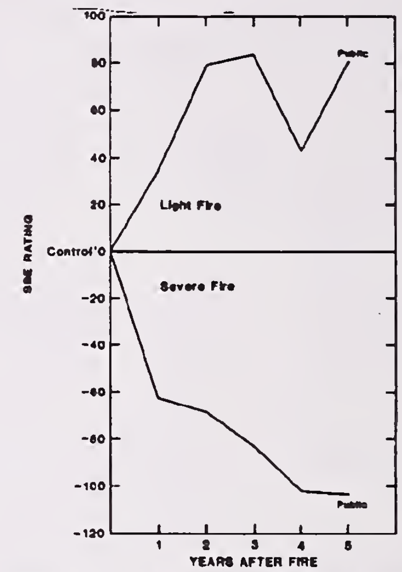
rioration, at least when the interviewing is not conducted during an air pollution event related to forest fire. However, the experience reported by forest managers, and certainly this year (1988) by the news media, is that **someone** gets upset when smoke from a forest fire invades his or her town. With the current state of knowledge, however, we don't really know which segments of the public respond negatively to smoke impacts, other than concessionaires and other businesses reporting economic losses. Public hearings and workshops concerning prescribed burning receive much positive response. The unanswered question is whether these same people reverse their positions when they are actually experiencing smoke from a fire or if these different responses come from two different segments of the public. Taylor and Mutch (1986) suggested a "quick response survey" project to answer this question, but the suggestion has not yet been supported. Yellowstone National Park is starting a survey for 1989 concerning public response to the smoke and fire impacts of the 1988 fires. This project may be able to reveal who, among the public, are truly concerned about forest fire smoke.

Long-Term Effects of Fire

Research on the long-term effects of fire on scenic quality and on the acceptability of burn areas for recreational use is fairly recent. In their study plan for "Evaluation of public response to use of prescribed fire in recreational land management," Cortner et al. (1979) stated "Little previous research has systematically investigated public knowledge and attitudes toward fire management (in a recreational context)." Perkins (1971) reported on the effects of prescribed burning on outdoor recreation to a prescribed burning symposium in the southeastern United States. Anderson et al. (1982) found that a rapid scenic quality recovery may well follow prescribed light fire in a southwestern ponderosa pine forest. Taylor and Daniel (1982) conducted the first experiment that directly compared the residual effects of light, prescribed fire with those of severe, wild fire. This was done in the southwestern ponderosa pine forest type and included evaluation of the effects of fire on the acceptability of these areas for recreational use, as well as for scenic quality.

Historically, the relation between fire and recreation has been assumed to parallel some other interaction dimension, generally the relation between fire and scenic quality. Rudolf (1967) specifically equated proper management for recreation with proper management for visual quality in discussing silviculture for recreation area management. Perkins (1971) assumed that the effects of prescribed fire on outdoor recreation would parallel the effects on plant and animal species composition. He argued that because most recreation activities are dependent upon the presence of specific species (for hunting, fishing, bird-watching) or upon species diversity (for nature study, photography), forestry practices that enhance appropriate species composition must, by definition, enhance recreation. By this rationale hunting, camping, picnicking, hiking, bird-watching, and outdoor photography were all assumed to benefit from prescribed burning. Wagar (1974) dealt specifically with aesthetic amenity values and effects in discussing recreational and visual quality considerations of forest residues management. The visual aesthetic dimensions that Wagar considered included naturalness, imageability, legibility, texture, harmony, scale, and order. Only through the dimension of "passability, the openness to human passage," was an outdoor recreation dimension given direct consideration.

It is generally assumed that there is a relation between forest management practices and aesthetic quality and between scenic quality and recreation. A properly managed forest is a thing of beauty to a silviculturalist, but this may not necessarily be true for the general public. It has taken many years for the wildlife biology community to convince foresters that optimum silvicultural management



*Control level represents rating of areas having had no fires in 100 years.

Figure 5.—Scenic beauty estimations of forest areas following fire (Taylor and Daniel 1982).

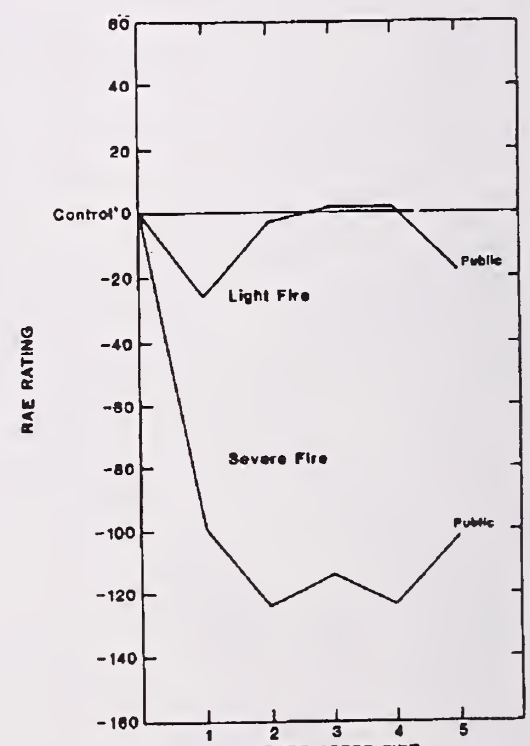


Figure 6.—Recreation acceptability estimations of forest areas following fire (aggregated data) (Taylor and Daniel 1982).

does not necessarily provide optimum wildlife habitat. It may take an equally long time to demonstrate what the real relations are between fire management and scenic quality; between the effects of fire on aesthetics and recreational acceptability.

One dimension of the Taylor and Daniel (1982) study was a direct comparison between the effects of fire on scenic beauty estimations and on recreational acceptability estimations. To test these effects, ponderosa pine forests in Arizona were photographed. Areas selected and photographed showed 1 to 5 years of vegetative recovery from two intensities of fire: light intensity, prescribed fire, and severe wildfire. A control ponderosa pine forest area was used that had had no fire in the past 100 years. Respondents, selected from public and church groups in Tucson, were asked to independently rate two sets of scenes, depicted in 35-mm slides: one set for recreational acceptability and one for scenic quality. Both

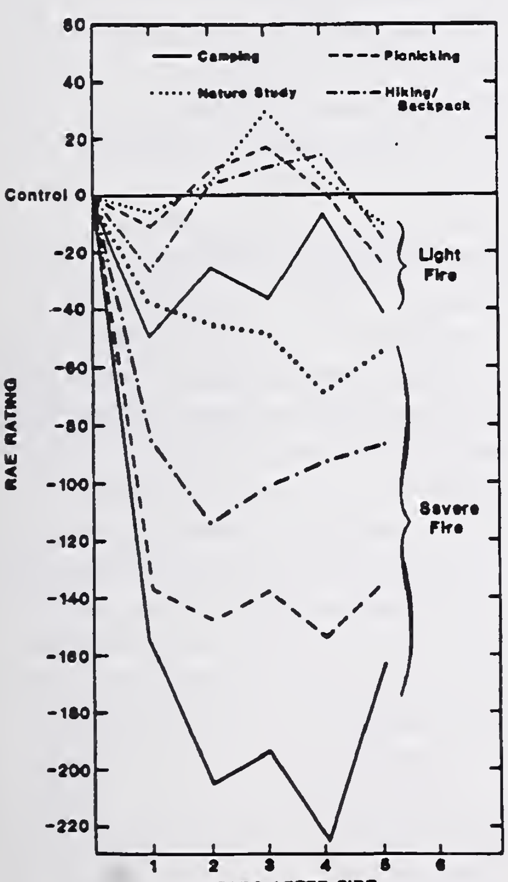


Figure 7.—Recreation acceptability estimations of forest areas following fire (by recreation activity) (Taylor and Daniel 1982).

evaluations were patterned after the Scenic Beauty Estimation [SBE] methodology established by Daniel and Boster (1976). For the recreational acceptability estimation [RAE], each participant selected the outdoor recreation activity most preferred for the kind of forest areas shown, then rated each scene for "how good that area would be for that recreation activity."

In general, the respondents' estimations of the scenic quality and recreational acceptability of the forest burn areas shown responded quite strongly to the effects of different fire intensities. Perceived scenic beauty (fig. 5) for light fire was improved, especially by the second year after the fire, as compared to the scenic beauty estimation for the control. Severe fire seriously eroded scenic beauty, with a general worsening trend over at least the 5-year test period following the fire. It may be that this worsening condition is a result of weeds and shrub growth as early post-fire recovery stages.

Recreational acceptability (fig. 6) also showed a differential response to light and severe fire intensities. In this case, however, light fire effected little change from the control, no-fire forest. Recreation acceptability response to severe fire dropped immediately and stayed there, rather than showing a decreasing trend. A comparison of Figures 5 and 6 shows that recreational acceptability estimations do not directly parallel the scenic beauty estimations. The curves are different—especially the response to light fire effects in comparison to the control, no-fire condition—and the spread between light fire and severe fire response is greater for scenic beauty than for recreational acceptability. From this it should be apparent that for the general public, evaluating the effects of light and severe fire in the southwestern United States, scenic beauty is not the same thing as acceptability for recreation. Indeed, in this context, scenic quality cannot even be used as a reasonable surrogate for recreational acceptability; one could not accurately decide, despite the intuitive logic, that an area that has been aesthetically improved through the use of prescribed burning is necessarily improved for recreational use.

This discussion is based on comparison of scenic beauty with the aggregated recreational acceptability estimations. What of the response to fire by separate recreational activity groups? Do hikers and campers differ in their estimations of recreational acceptability of forest burn areas? Figure 7 shows the recreational acceptability results from the Taylor and Daniel (1984) study by recreation activity. These data should be accepted with some caution: respondents in this survey self-selected their favorite recreation activity to evaluate vis-a-vis fire. The subsamples, therefore, could not be stated to differ from each other only on the dimension of recreational choice selected. Nevertheless, the pattern of response by recreation activity is sufficiently clear to warrant at least preliminary conclusions.

Of the four recreational activities selected by respondents in Taylor and Daniel's (1982) test, camping was most sensitive to fire effects. The negative effects of severe wildfire on camping are about twice the magnitude of the negative effects on scenic beauty. Camping even showed some negative response to light, prescribed burning effects. Next most sensitive to fire effects was picnicking, which showed a severe negative response to wildfire effects but very little response to light fire. Hiking/Backpacking showed about the same degree of negative reaction to severe wildfire as the scenic beauty estimations, but again little positive or negative impact from light fire. Nature study was least affected by severe fire and may have had a slightly positive response to light fire, although certainly not significant. Anecdotal evidence, volunteered by respondents in this survey, gives some

explanation of the seemingly anomalous response by those selecting nature study. "It may look awful," one woman said, "but it would be an excellent place to study wildlife."

These recreational acceptability responses demonstrate not only that recreational acceptability is a different phenomenon than scenic beauty, but that acceptability of forest burn areas for recreational use varies significantly according to the severity of the fire and from one outdoor recreation activity to another. Thus, decisions as to whether to prescribe burn or let a naturally ignited fire burn in a specific forest area will have to depend not only on whether there is outdoor recreation use of the area, but on what types of outdoor recreation occur. These results are preliminary; further research is required to statistically verify just how responses to fire effects vary from one form of outdoor recreation to another.

Implications for Management

Implications for these various findings, for both fire management and recreation management in southwestern forests, are set here in the context of information and education programs. In a forthcoming chapter on public attitudes and perceptions about prescribed burning in the Pacific Northwest, Shelby and Speaker (in draft) identified the following "key elements" for a successful Information and Education program about fire management:

1. A long-term effort to inform the public about the natural role of fire in undisturbed ecosystems;
2. A strong consensus among forest managers and concerned user groups about the correct use and beneficial effects of prescribed fire;
3. Public perception that the information is scientifically sound and not stemming from an interest group with a biased position; and
4. Adequate treatment of specific public concerns related to the use of fire, including the risks of prescription fires getting out of control, smoke intrusion into populated areas and related effects on public health, potential health hazards of burning chemically treated sites, and aesthetic impacts.

Two very important axioms to consider for public information programs are that (1) information exchange is only effective as a two-way process and (2) educating the public will not necessarily cause them to believe as you do. Unfortunately, there is a strong tendency, across a wide spectrum of resource professionals, to commit the two cardinal errors that oppose these axioms. "We have come a long way in our understanding of fire ecology. The public, ignorant of these advances, finds our policy changes very hard to accept. If we educate the public about what we've learned, we will garner the public support needed for our new fire management programs." Note that this approach does not assess public knowledge, but assumes ignorance on the part of the public. Second, it assumes that the public will come around to our way of thinking if they are educated to our relatively new-found knowledge. We should intuitively recognize the danger and lack of foundation of the second fallacy—that once educated, people will believe as we do. Certainly, some of the most effective opposition to resource management practices have come from the most knowledgeable segments of the public—from the forest products industry on one hand and from environmental action organizations on the other.

Information and education programs can be thought of as cyclical processes. First, inform ourselves about the public, then set about the public education that is appropriate. In summary, what is known about public beliefs and attitudes concerning fire? Growing acceptance and sophistication characterize public attitudes toward current fire management practices. Prescribed burning is generally well accepted; fires started by human carelessness or by lightning are not. As the nature and degree of control of a fire are better understood by the public, there is a tendency to be more accepting of managers' decisions. However, one area of misinformation, mentally associated with the rather strong public rejection of allowing lightning-caused fires to burn, should be of concern to the fire manager. This is the usual origin of forest fires in the interior West. The implications for prescription burns with unplanned ignitions are important. Only YOU can start forest fires, Smokey not withstanding.

The average public ability to correctly respond to fire questions is increasing. Knowledge has increased significantly for questions regarding the relation of fire to nutrient availability, the control of insects and disease, and the relation of fire suppression to changes in community structure and to the intensity of future wildfires. However, knowledge remains low about the average size of forest fires before suppression activities began, the number of animals killed in forest fires, the lightning-origin of most fires in western regions, and the effects of suppression on wildfire intensity and animal habitat. Knowledge about fire suppression's relation to future fire intensity remained relatively low in McCool and Stankey's (1986) survey, although it increased from 1971 levels.

The relation between knowledge levels and factors that explain acceptance of fire management practices, as shown in table 2, should be considered. Understanding some of the beneficial effects fire can have on forest ecosystems is important in ac-

cepting various types of fire. This connection is intuitively logical. Knowing that lightning is the usual cause of forest fires in many western forest types also is significant in people's acceptance of a variety of fires, but this knowledge is not very widespread. Knowing that most fires in forest ecosystems are small and that most animals are able to escape from wildfires are important in acceptance of fires that are not specifically designated as set and controlled by fire managers. Again, a limited segment of the public is aware of these factors.

These results show mixed public knowledge and beliefs about fire behavior and effects. Some parameters are understood, some are not known, and still others are clearly misunderstood. Examining public knowledge can prevent mistakes such as talking down to an audience that already understands the message, going past issues about which the audience is confused, or missing areas where the audience is quite confidently wrong in their beliefs. A public information program that is not based on surveying public knowledge could make all these mistakes simultaneously.

Fire Managers' Decision Behavior

If I insist that resource managers must study their audience before prescribing education programs, then it should be incumbent upon me to have studied resource fire managers before presenting this paper. Indeed, I have participated, over the past few years, in a survey of USDA Forest Service, Fire Managers' fire-risk decision behavior (Taylor et al. 1987, 1988). This survey was designed to find out what factors weighed heavily on fire managers as they faced decisions involving risk. The survey design was kept as simple as possible. Fire managers read scenarios describing a fire-decision situation for each of three contexts: responding to an escaped wildfire, deciding on setting a prescribed burn, and long-range fire-budget planning. In each case they were required to decide: the level of attack on the wildfire, whether to set the prescribed burn, or the level of budget or risk to accept. Then, the degrees to which various decision factors influenced those decisions were assessed.

Pertinent to the present discussion, resource issues had profound effects on fire managers' decisions. Indeed, resource issues ranked nearly even with safety issues in influencing fire decisions. However, not all resources are given equal weight, despite legislation demanding multiple-use management. Threats to timber received the highest decision influence ratings by fire managers for both escaped wildfire and prescribed fire situations and was ranked fourth for long-range fire-budget planning. "The chance of a catastrophic fire is likely to be reduced" was rated highest for long-range planning. The fairly new national policy of balancing the value of resources at risk with the costs of fire activities received the second-highest fire-decision influence rating overall. The item, "More money could be spent than is justified by the resource values" at risk was rated second for both escaped wildfire and long-range planning decisions, third for prescribed burning.

Where does recreation fit into this scheme? The item "recreation opportunities could be lost" was rated fifth, out of 18 factors, for the escaped wildfire decision context. However, protection of recreation resources did not fare so well in prescribed burning or long-range fire budget planning contexts. Reduction of recreation opportunities was ranked 16th, of 19 factors, for prescribed fire decisions and 17th, of 18, long-range planning decision factors. Similar to George Orwell's "Animal Farm" (1954), some resources are "more equal than others."

A comparison of the priority given recreation resources by these fire managers with those of the general public, as reported by Zwolinski et al. (1983), suggests that some realignment is necessary. The public sample rated outdoor recreation third in importance among forest resources in the Southwest, above lumber, livestock feed, firewood, and hunting. Concern about the effects of fire on recreation areas also was rated fairly high, with nearly 40% of the respondents reporting that they would be "extremely concerned" that "recreation areas will be destroyed" when they see evidence of a forest fire. As the fire risk decision data (Taylor et al. 1988) indicate, a number of resource managers are retaining resource priorities from past decades, giving preferential treatment to timber over important emerging public values for recreational use of forest resources.

Seldom do resource managers weigh recreational values higher than such tangible commodities as timber production or livestock grazing. Even less heavily are weighed the subtle differences among different recreational expectations in resource or fire management decisions. However, mechanisms for incorporating these subtleties, such as the Recreation Opportunity Spectrum (Driver and Brown 1975) are becoming available for use by the resource manager.

Evidence is mounting, from the various national outdoor recreation surveys that have been conducted over the past few decades, that demand for outdoor recreation activities is continuing to increase. Clearly, it is important for the resource fire manager to align his or her priorities to match those of the constituent public. Recreation is of prime concern to the American public and must be given high priority weighting by natural resource managers in making decisions about fire.

Using Fire as a Management Tool in Southwestern Ponderosa Pine[1]

Michael G. Harrington and Stephen S. Sackett[2]

Abstract.—Fire suppression and livestock grazing over the last century are responsible for current forest conditions of extensive stand stagnation, uncommonly high fuel accumulation, and general low productivity. From research on fire effects, knowledge is now available for using fire to reduce fuel hazards, thin dense thickets, and provide sites for ponderosa pine (*Pinus ponderosa* Laws.) establishment.

CHANGING CHARACTERISTICS OF SOUTHWESTERN PONDEROSA PINE FORESTS

The ponderosa pine (*Pinus ponderosa* Laws.) forests of the Southwest have gone through extensive structural and compositional changes in the last century. Numerous references document the open, park-like appearance of historic ponderosa pine stands (Biswell et al. 1973, Brown and Davis 1973, Cooper 1960), where herbaceous vegetation was vigorous and abundant. Fires were a regular feature of these forests, burning the light surface fuels at intervals usually averaging less than 10 years and as often as every 2 years (Dieterich 1980, Weaver 1951). The frequency of these fires resulted from the continuity of grass and pine needle fuels, the high incidence of lightning, and the warm, dry weather common to the Southwest. Light surface fuels built up sufficiently with the rapid resprouting of grasses and the annual pine needle cast. Large, woody fuels, which fall infrequently, rarely accumulated over extensive areas. When single or small groups of trees fell, they were generally consumed by subsequent fires, creating a mineral soil seedbed and reducing grass competition in microsites, favoring ponderosa pine seedling establishment (Cooper 1960). These circumstances created an uneven-aged stand structure composed of small, relatively even-aged groups.

Change began in the southwestern ponderosa pine forests during extensive livestock grazing in the late 19th century (Faulk 1970). As grazing intensified herbaceous vegetation could not respond, and its coverage declined drastically. This decline led to two subsequent changes: reduced fire spread because of the decrease in fine fuels, and an eventual increase in ponderosa pine regeneration because of reduced competition and fire mortality, and more mineral seedbeds (Cooper 1960). Beginning in the early 1900's, forestry practices, including fire control, further reduced the spread of inevitable fires, leading to unprecedented fuel accumulations and stagnation of seedling and sapling thickets.

These human-induced changes have resulted in ponderosa pine forests that have little similarity to the presettlement forests. In the uncut or lightly harvested stands, old-growth trees still stand. The open structure is gone, however, as dense sapling thickets and small pole groups have developed in the understory. Stand stagnation has been reported on many sites (Cooper 1960, Schubert 1974), and persists where natural or artificial thinning has not taken place. In addition to stand changes, 75 to 100 years of general fire absence has also led to uncharacteristically large accumulations of surface and ground fuels (Kallander 1969).

Sackett (1979) reported average loadings of naturally created fuels at 22 tons per acre (range 8-48 tons per acre) for 62 southwestern ponderosa pine stands. Harrington (1982) verified the heavy fuel loadings, with an average of 34 tons per acre in southeastern Arizona. Another formerly uncommon feature is the abundance of large, woody fuels, averaging about 8 tons per acre. Much of these down, woody fuels have accumulated in sapling thickets, creating an even more severe hazard. A final characteristic of current southwestern ponderosa pine stands is the sparse understory vegetation, created from thick forest floor layers and dense pine canopies, that resulted from fire suppression (Arnold 1950).

The changes that have taken place primarily within the last century have created several undesirable conditions in the ponderosa pine forests of the Southwest. The extreme fuel hazard is probably most apparent. The combination of heavy forest floor fuel loadings and dense sapling thickets coupled with the normally

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[2]Research foresters, USDA Forest Service. Harrington is with the Intermountain Fire Sciences Laboratory, Intermountain Forest and Range Experiment Station in Missoula, MT; Sackett is at the Forest Fire Laboratory, Pacific Southwest Forest and Range Experiment Station in Riverside, CA.

dry climate and frequent lightning- and human-caused ignition potential result in a severe wildfire threat (Biswell et al. 1973, Harrington 1982). Additionally, trees of all sizes have generally poor vigor and reduced growth rates (Cooper 1960, Weaver 1951). This condition is likely due to the reduced availability of soil moisture caused by intense competition and by moisture retention in the thick forest floor (Clary and Ffolliott 1969). The thick forest floor also indicates that soil nutrients, especially nitrogen, may be limiting because they are bound in unavailable forms (Covington and Sackett 1984).

In the past, pine regeneration generally developed in openings in the stand after fire had produced a receptive seedbed. Now, long after tree mortality and the creation of openings, a poor quality, organic seedbed remains, allowing little opportunity for establishment of pine seedlings. Also, forage production for wildlife and livestock is commonly minimal, because of severe competition with trees and the physical effect of the deep forest floor (Biswell 1972, Clary et al. 1968).

Ponderosa pine is also found with several other conifers in the higher elevation mixed-conifer forests. Fire occurred less frequently in these more mesic forests. A 22-year average interval was found in northeastern Arizona (Dieterich 1983), but its impact was still important for site preparation, species selection, disease and insect distribution, and nutrient cycling (Jones 1974). Little documentation of fire effects and burning prescriptions is available for southwestern mixed-conifer forests, therefore, this subject will not be addressed here.

This paper addresses the use of prescribed fire in southwestern ponderosa pine forests. From research and observations, recommendations are made to apply fire to reduce fuel hazards, thin dense sapling thickets, and provide conditions for natural regeneration in natural forest stands.

SOUTHWESTERN PONDEROSA PINE FUELS

The deteriorating and hazardous forest conditions described above have contributed to numerous severe wildfires in Arizona and New Mexico. Several examples include the 1951 Escudilla Mountain Fire (19,000 acres) and Gila Black Ridge Fire (40,000 acres), the 1956 Duddly Lake Fire (21,000 acres), the 1971 Carrizo Fire (57,000 acres), and the 1977 La Mesa Fire (15,000 acres) (Biswell et al. 1973, Cooper 1960, USDA 1977). The need to alleviate the hazards was recognized years ago, but widespread use of fire under specified conditions to create designated results has been adopted slowly. Reasons for this reluctance include a shortage of guidelines for prescription burning and insufficient information on fire effects.

Fire Effects on Fuels

The use of prescribed fire has increased in recent years. Some of the earliest, extensive prescribed burning took place on the Fort Apache Indian Reservation in Arizona with about 3,000 acres burned in the late 1940's (Kallander 1969). From 1950 to 1970, over 300,000 acres were burned, primarily for hazardous fuel reduction. The effectiveness of this burning program in reducing size and severity of subsequent wildfires has been documented (Biswell et al. 1973, Knorr 1963). This burning operation used the cool, dry conditions encountered in late fall to moderate fire behavior for initial fuel reduction burns. Burning was not to begin until after November 1, but dry litter and duff layers were preferred (Kallander 1969). These fires were strategically ignited, then allowed to burn, unattended, over vast acreages. In a series of three burns in 1950, this procedure was used to burn portions of 65,000 acres (Weaver 1952). Forest floor fuel loadings were reduced by 55% and dead woody fuels were reduced by 64% to 80%. Emphasis was placed on the consumption of thousands of snags and windfalls, which greatly lessened the fire hazard. The effect of this burning operation was evident the following year by a dramatic reduction in the number of wildfires and in the acreage burned (Weaver 1952).

In a well documented study in 1950, six small plots were burned under two sets of fuel moisture and weather conditions to determine fuel consumption and tree responses (Gaines et al. 1958). A low intensity burn was set at the end of September, and a high intensity burn was set in early October with higher air temperatures and lower fuel moistures, humidities, and winds. The September burn resulted in a 57% reduction in surface, ground, and aerial fuels. The October burn consumed more fuel, but also created new fuel by killing or damaging numerous trees. When the newly deadened fuels were added to the residual fuels, only a 15% reduction was realized.

Another large scale, fuel-reduction burn on the Fort Apache Reservation was conducted in November 1956 under cool, clear days with a moderately high drought index and low rate-of-spread index (Lindenmuth 1960). Fires were ignited at specific locations, but allowed to burn unchecked for 33 days within project boundaries. Fire effects were undocumented in fall 1957 (Lindenmuth 1962). Fuel reduction was deemed unsatisfactory because fuel consumption on 75% of the area was only minimal. In addition, a large portion of the area that had good fuel reduction also had extensive tree mortality or injury.

In central Arizona, a small scale research burn was conducted on two distinct sites. The first had 75% more fuel by depth and weight, and 85% greater overstory basal area than the second (Davis et al. 1968). Under similar fuel moisture and weather conditions, more fuel was consumed

on the site with the most fuel. However, 2 years after the burns, the net fuel change including consumption by fire and litter accumulation demonstrated that the site with less initial fuel had a 37% reduction compared with a 23% reduction on the other site. This indicates more damage was done on the site with more fuels, resulting in greater fuel accumulation.

In a more recent study in uncut ponderosa pine in north-central Arizona, the importance of fuel moisture on consumption was seen (Sackett 1980). Two similar sites at Fort Valley and Long Valley Experimental Forests, maintained by the Rocky Mountain Forest and Range Experiment Station, were burned in fall under contrasting moisture and weather conditions. Surface litter had similar moisture on both sites, but the humus layers (lower duff) differed 10% to 15%. Since about 80% of the forest floor weight is in the ground fuels, the ground fuel moisture is most influential in total fuel consumption. Over 60% of the ground fuels and about 70% of the large woody fuels were consumed in the drier burn compared with only about 40% of both fuel groups in the wetter burn.

The importance of humus moisture was demonstrated further by a prediction equation for forest floor reduction developed from a series of summer burns in southeastern Arizona (Harrington 1987). In the equation,

$$FR = 37.4 - 0.8\,HM + 21.2\,FD \quad (1)$$

humus moisture (HM) and preburn forest floor depth (FD) were highly correlated with percent forest floor loading (FR) reduction ($r^2 = 0.91$). Tree density also had a strong effect, with less fuel reduction occurring in denser stands.

Equation [1] clearly indicates that the percentage of forest floor (FR) consumed increases as the preburn amount (FD) increases. This result was confirmed by the results of an October 1982 initial fuel reduction burn at the Fort Valley Experimental Forest (Covington and Sackett in press). Under low humidities (15% to 24%) and moderate air temperatures (52° to 67° F), nine plots with differing stand structure were burned. Surface fuel moisture ranged from 7% to 10%, and humus moisture ranged from 12% to 20%. In stagnated sapling thickets, about 34% of their 12 ton per acre loading was consumed, 52% of the 16 tons per acre in the pole stands was consumed, and 89% of the 55 tons per acre was burned in the mature, yellow pine stands. A high, positive correlation was found between percent forest floor consumed and preburn loading ($r^2 = 0.89$).

Although there are valuable fuel hazard, nutrient, and regeneration benefits derived from the consumption of heavy forest floors, there are also liabilities. Consumption of large quantities of fuel generates large amounts of heat energy. Studies at the Fort Valley and Long Valley Experimental Forests and Sequoia-Kings Canyon National Park show very high mineral soil temperatures during burning. Lethal temperatures have been measured on many sites deeper than 12 inches into the mineral soil. More than 35% of the old-growth ponderosa pines, which survived numerous presettlement fires, have died at Fort Valley as a result of the first burns in 100 years.

Fires burning under these old-growth pines are unspectacular, consuming only the litter in the flaming front. Most of the forest floor is consumed by smoldering combustion, which may take 72 hours to complete. Burning for this length of time can result in either temperatures exceeding 140° F, which cause instant cambium or root death, or lower temperatures for longer durations, which also cause tissue death.

Fuel loadings under these old pines at Fort Valley ranged from 41 to 86 tons per acre, and consumption was always greater than 85%, regardless of humus moisture up to 90%. In 13 of 14 cases, 140° F was reached at 2 inches below the soil surface. At 8-inch depths, temperatures frequently exceeded 100° F for at least 6 hours, with some reaching 140° F. On a burn at the Long Valley Experimental Forest under wetter conditions, 69% of the 45.5 tons per acre of forest floor was consumed. Temperatures failed to reach 100° F at the 8-inch soil depth, but ranged from 140° to 210° F at the 2- to 4-inch depth.

Live ladder fuels, which also add to the hazard, need particular attention. These fuels can take the fire from ground level into the overstory crowns. The vertical fuel continuity can be broken by a reduction in ladder fuels. Some of the important live fuels are medium-sized shrubs which are easily top-killed by fire but generally resprout. Examples include Gambel oak *(Quercus gambelii)*, silverleaf oak *(Q. hypoleucoides)*, and wavy leaf oak *(Q. invaginata)*. Gambel oak can be temporarily reduced by prescribed burning (Harrington 1985). Low crowns of ponderosa saplings also increase crown fire threat. Scorching the lower foliage is effective in relieving this hazard, but the burning procedure is difficult and requires experience (Harrington 1981, Kallander 1969).

Understory burning in southwestern ponderosa pine can greatly, but only temporarily, reduce the fuel hazard (Harrington 1981, Sackett 1980). Consumption of the litter layer lessens ignitability and rate-of-spread potential. As more of the duff, ladder fuels, and large logs are consumed, a reduction in potential fire intensity, total energy release, and resistance to control are realized. Truesdell (1969) reported a decrease in wildfire size up to 7 years after prescribed burning on the Fort Apache Indian Reservation and a 3-year reduction in suppression costs after burning on the Hualapai Indian Reservation. As mentioned, the fuel hazard reduction is only temporary as 0.6 to 1.8 tons per acre of needle litter can be cast annually depending on tree density and site productivity (Davis et al.

1968, Sackett 1980). Fuel often accumulates rapidly to hazardous levels after initial fuel reduction burns. As needles from scorched trees fall, the amount of surface litter can actually become greater than preburn levels. This litter increase is the result of the inevitable tree injury caused by consumption of the unnaturally great fuel quantities in dense stands. Therefore, reburns are essential to remove these fire-created fuels and generally maintain low fuel hazard, even when initial burns are effective (Harrington 1981, Sackett 1980).

Reducing Fuels with Prescribed Fire

A lot has been learned within the last 50 years about the use of fire in the Southwest. Many fire experts have developed their skills primarily through personal experience, learning from failures as well as successes. This type of knowledge is difficult to pass on to less experienced individuals. However, there is now enough documentation of research and operational burns to provide general guidance for fire prescription and effects. Unique combinations of stand, fuels, vegetation, and terrain may preclude the use of the following prescriptions and effects information. Therefore, we recommend a thorough assessment of site characteristics. A generalized set of fire prescription parameters was derived from the prescribed burns discussed earlier.

Season

In forested sites where fire has been absent for decades, the initial fuel reduction burns should be conducted in the fall or early spring when temperatures and humidities are moderate. Fall burning can begin as early as mid-September and can continue in some years into December.

Weather Parameters

The following prescription parameters are the primary variables that determine whether a fire will burn successfully, hazardously, or not at all. On sites requiring reduction of natural fuels, maximum daytime air temperatures should be between 50° and 75° F. Below 50° F, moderately dry fuels (9% to 12% moisture) burn poorly and above 80° F extensive overstory crown scorching is likely. Minimum relative humidities should not drop below 20% or exceed 40%. Fuels subjected to a series of low humidity days become hazardously dry. Also, very low humidities are frequently accompanied by temperatures above 80° F. If minimum humidity exceeds 40%, light surface fuels are generally too moist to burn well. Windspeed at flame height should be between 3 and 8 miles per hour. Slope effects can compensate for lack of wind. A fire burning with little or no wind and no effective slope either will not spread well or will cause extensive crown heating, if fuels are dry. Windspeeds greater than 10 miles per hour can result in erratic fire behavior. Surface pine needles ideally should contain 5 to 12% moisture. Below 5%, ignition and rates-of-spread are too rapid, and above 12%, burning is patchy and incomplete with slow rates-of-spread.

Not all combinations within the range of temperatures, humidities, windspeeds, and fuel moisture described above are safe and effective. For example, if burning conditions are approaching the upper temperature and windspeed limits and the lower humidity and fuel moisture limits, a very intense, rapidly spreading fire will result. However, experienced burners can use the upper limits of one parameter to make up for a deficiency in another. For example, a combination which provides good burning conditions is low humidity (15% to 20%) and low temperatures (40° to 50° F). These situations do occur in late fall throughout the Southwest.

Because damp, cool fall weather often results in poor burning conditions, summer burning during the monsoon season has been studied as a successful alternative (Harrington 1981, 1987). The amount of drying that follows fuel-saturating rains will determine fire behavior and fuel consumption. Using the same prescription ranges during the summer rainy season should permit successful fuel reduction burns. More attention to air temperature limits and erratic winds is needed, however.

Follow-up

Maintenance burning is necessary to keep the reccurring fuel hazard to a minimum (Davis et al. 1968, Gaines et al. 1958, Harrington 1981, and Sackett 1980). Since most of the light, fire-created fuels accumulate within 3 years of burning, we recommend a repeat burn within that period. Generally, repeat burns in light, needle fuels are easily managed. The window of burning season and ambient conditions is broader than for initial burns, with warmer, drier, windier situations being advantageous to the conduct of the burn (Harrington 1985). Air temperatures should range between 55° and 85°F, humidities from 15% to 40%, windspeeds from 5 to 12 miles per hour, and litter moisture from 5% to 10%. After the second or third burn, annual litter accumulation should return to a level relative to natural attrition. From this point, burning need only be conducted at intervals of about 7 to 10 years to maintain a low hazard.

If a reduction in sprouting shrubs is a major management goal for fuel and competition reduction, then a distinct program of repeat burning is needed. For Gambel oak management, we suggest an initial fuel reduction burn in fall followed by 2 or 3 mid-August burns, 2 years apart (Harrington 1985).

Predicting Fuel Consumption

The ability to predict fuel consumption from prescribed burning would be valuable to forest managers. If too little fuel is removed, the hazard might not be relieved; if too much fuel is consumed, then tree mortality might be excessive, and site quality might be compromised. Equation [1] has not been extensively tested, but should work reasonably well in stands with characteristics within the following ranges: preburn fuel loading = 25 to 40 tons per acre, large woody fuel loading = 3 to 15 tons per acre, and stand density = 800 to 2200 trees per acre (Harrington 1987). The estimation of fuel reduction was fairly accurate for fires reported by Sackett (1980). However, it was highly over estimated for fires reported by Davis et al. (1968) and Harrington (1985), because fuel loadings and stand densities were one-half or less of those used in the equation development.

Reducing Logging Slash

Reduction of fuels from silvicultural activities is also important for lessening the chance of severe wildfire in residual stands, especially in ecosystems where the wildfire potential is so great. However, little documentation exists concerning effective combinations of cutting methods and fire. Buck (1971) proposed burn prescription parameters and techniques that have worked well in reducing logging slash while causing acceptable tree damage.

During harvesting, generally a large amount of logging slash is added to the existing natural fuel component discussed earlier. This added slash creates an extreme hazard and leaves the fuel manager or silviculturalist with a complex condition to attempt to relieve. An informal proposal has been suggested in which a preharvest burn would be conducted under conditions described above to reduce natural fuels, followed by a postharvest burn to reduce activity-generated fuels and aid site preparation.

STAND DENSITY IN SOUTHWESTERN PONDEROSA PINE FORESTS

Besides reducing the wildfire hazard, thinning of such stands releases the residual trees, allowing faster growth (Schubert 1971). Domestic and wild animals also benefit from thinning of dense stands. Forage production is increased to a higher level by reducing the basal area in typical ponderosa stands (Clary and Ffolliott 1966, Jameson 1968).

Forest visitors find the dense thickets of reproduction uninviting. Access to the forest is often inhibited and esthetic values are reduced. These dense stands do not indicate a healthy ecosystem. Treatment measures that are silviculturally acceptable and economically sound are needed to improve the situation.

These "dog hair" thickets may be modified for various reasons in a variety of ways. Sometimes an entire thicket should be eradicated when infected by disease or insects, or when release potential is minimal. Fuel breaks are sometimes thinned to a low basal area for fire suppression. Stand density in travel influence zones is also reduced to improve visibility. Thinning as a silvicultural treatment is generally quite limited by cost, but some reduction of tree density could benefit over-dense stands on more than 4 million acres of southwestern ponderosa pine land (Schubert 1974).

Many of these stands need thinning to provide for more productive forests. Mechanical or hand thinning are probably the most common methods used, but they do not appear to be economically feasible over extensive areas. In addition, the hazard created by extensive thinning operations would encourage insect outbreaks and make fire protection even more difficult. Obviously any one treatment is not the answer. The problem must be spread over a number of methods, depending on circumstances and situations present.

Effects of Fire on Stand Density

There has been limited research on the use of fire to eliminate the fuels produced by thinning. One documented case of successfully thinning slash used high, green fuel moisture to minimize ponderosa pine mortality in Oregon (Smith et al. 1983). Even less information is available on the use of fire to accomplish thinning. Fire is not a very selective thinning tool, producing a rather unpredictable, patchy residual stand. However, most studies dealing with fire as a thinning tool have lacked a long-range process to accomplish the objective. A number of fires are required to reduce fuels, change the understory, and overcome the changes caused by fire exclusion.

Thinning by fire was a natural process in ponderosa pine before settlement. The degree of thinning is dependent on the quantity of fuel on the ground (Cooper 1961). The more dense the thicket, the more fuel, and the more intense the fire; thus resembling a self-regulating feedback mechanism governed essentially by stand density.

A number of investigators have dealt with the use of fire as a thinning tool—an emulation of the "natural" pine processes. Weaver (1947) reported on a comparative study of stand conditions in an area burned in September 1914 and an adjacent area not burned, on the Colville Indian Reservation in Washington. Thirty years after the fire, the 40-year-old, fire-thinned stand had substantially fewer stems per acre, greater height, and larger diameters than the adjacent unburned stand. The conclusion was that fire was an effective thinning tool. Tests on the Fort Apache

Indian Reservation using prescribed fire as a thinning tool showed that thinning was spotty from fall fires, but the prescribed fire did a "reasonably effective and conservative job of thinning" (Weaver 1952). Gaines et al. (1958) wrote a supplement to Weaver's report providing data that more or less supported the previous observations. Their conclusions were not as optimistic because of the injury to the commercial overstory, and noted the need for additional information. Wooldridge and Weaver (1965) reporting on a prescribed fire on the Colville Indian Reservation designed to thin dense sapling stands, concluded that prescribed fire was a rough and largely unpredictable thinning tool. The fire drastically reduced the number of stems, had no significant effect on diameter growth, and caused a slight net reduction in height growth.

Lindenmuth (1960) studied the effects of two prescribed fires in east-central Arizona near McNary and Maverick. The 1956 fires burned continuously for 33 days under a variety of topographic, fuel, and weather conditions. The fire released from competition 24.3% of the potential crop trees that needed releasing (a novel way of presenting thinning data). The fires also destroyed 10% of the potential crop trees and damaged an additional 7.4%. Lindenmuth concluded that these particular fires demonstrated an imperfect tool, and rightly so, since no specific thinning objective was intended, and because of the many varied conditions under which the fire burned.

In a study of prescribed fire in California ponderosa pine, Gordon (1967) minimized the benefits derived from fire. Using a limited data base (three fires), small areas, and severe burning conditions, he concluded that dense seedling and sapling groups would be completely killed by broadcast (prescribed) burning, and that fire is not a feasible tool for hazard reduction or thinning in eastside pine areas. Volumes of data were collected, but no regard was given to burning technique.

Ffolliott et al. (1977) reported an effective thinning response from an experimental prescribed fire near Flagstaff, Arizona. However, they concluded that basal area was not reduced enough for optimal growth of the residual stand. As stated before, one fire seldom corrects problems associated with 100 years of fire exclusion.

A study specifically designed to evaluate the effects of prescribed fire on thinning in ponderosa pine was conducted in western Montana (Henderson 1967). Small plots were set up on three different areas and burned under low, medium, and high fire-intensity days as defined by fire-danger rating levels. Thinning success was dependent on "close supervision of the fire intensity through manipulation" (Henderson 1967, p. 57). He used backing fires on "high intensity days," but could not adjust to any other technique because of the severity of conditions. At low fire danger, weather and fuel conditions were not adequate to sustain an effective fire spread. Medium conditions allowed for adjustment of burning technique to regulate intensity. In general, high intensity fires eliminated more stems than did low intensity fires, but no prescriptions could be developed from the few tests and the inherent variability involved.

In test fires on the Apache National Forest with logging slash, after shelterwood cutting, Buck (1971) observed overstory mortality. Although the prescribed fires were not designed as a thinning tool, they did accomplish some effective thinning from below. Eighty-three percent of the losses were in suppressed and intermediate trees. Most experiments in thinning with fire have had their emphasis in the ponderosa pine type primarily because of the large acreages that have not burned for so long, resulting in stand stagnation.

In some recent prescribed fires in Arizona designed to reduce fuel hazards, thinning was also an important benefit. In three distinct fires in the Santa Catalina Mountains, Harrington (1981) reported tree density reductions in the small or suppressed classes of 24%, 56%, and 43% in stands with preburn densities of about 2000 trees per acre. Percent tree reduction was positively correlated with amount of fuel reduction indicating that, with more research, degree of thinning could possibly be predicted.

Two other initial fuel reduction fires netted similar results. At the Fort Valley Experimental Forest, initial prescribed fires designed to reduce natural fuels (Sackett 1980) reduced the number of stagnated reproduction and sapling stems from an average of 1553 to 912 per acre. Small poles, many of which are also stagnated in thickets, were reduced from 192 to 156 stems per acre. In a companion study at the Long Valley Experimental Forest, fewer intense fires occurred due to a wet summer preceding the fall burn. An average of only 180 stems per acre was killed by the fire in the reproduction/sapling size classes. Virtually none of the small poles were killed outright.

No known studies and reports deal specifically with the problem of developing and using definitive burning techniques for thinning. Most references deal with a single fire as an answer to the problem. From experiences at the Fort Valley and Long Valley Experimental Forests, southwestern Colorado, and southern Arizona, quality of fuel rather than quantity, as Cooper (1961) suggested, appears to be more essential for producing high intensity fires in dense stands.

Work in surface fuel characteristics and experience with many prescribed fires indicate that only the newly cast needles (L layer) and upper portion of the fermentation layer (F) actually burn as flaming combustion in heavy, old forest floor accumulations. The lower F layer is matted and bound tightly together by

mycelium hyphae. As a result, the lower portion of the F layer acts more like a solid piece of fuel rather than as individual particles as in the L layer, and does not burn well.

In an undisturbed, well-developed forest floor, newly cast needles become rapidly colonized and bound by mycelium and therefore less burnable. When fire spreads over the forest floor, most of the fungi are destroyed. Needles that fall after a fire do not become readily infected and a much deeper layer of pure litter accumulates. When fire is applied a second time, all material cast since the initial fire is consumed (for up to at least 4 years' accumulation). Fire intensity, rate-of-spread, and flame length are much higher in response to the greatly increased available fuel. Hence, repeat burning in higher quality and quantity fuel does a better job of thinning stagnated stands.

Crown scorch and consumption kills trees and thins stands more effectively than bole girdling. Many of the stagnated sapling stands arose from the famous 1918 seed crop and subsequent regeneration. Although the trees have grown little in diameter, tree height and bark thickness have progressed normally through the past 70 years. The unusually thick bark prevents heat of low intensity fires from penetrating enough to kill trees. Subsequent burns in deep litter result in high intensity fires which cause extensive crown damage yet do not damage the bole.

Thinning Stands with Prescribed Fire

Manipulating the Fire

The most critical element in the use of fire as a thinning tool is the burner's ability to manipulate the fire or the fire environment or both to achieve slow-dissipating, high temperature air in the crowns. Manipulation of each fire can be achieved in a number of ways. Adjusting the direction of fire spread relative to wind direction is the most common technique. Heading or uphill fires move at a speed commensurate with windspeed creating longer flame lengths, greater speed, and higher intensities. Backing fires, moving against the wind (or down hill), progress very slowly with short flame lengths and low intensities. Back fires seldom thin stands.

Using ignition techniques that interact with one another is probably the most effective way to thin stands. For example, a head fire and a back fire coming together create a vertical heat rise that is slow to dissipate and concentrates the heat in the crowns. The same effect can be accomplished by lighting a spot fire in the center of a thicket followed by a ring fire around the thicket. This technique generally eliminates the center, but leaves the outer ring of trees. Merging flank fires have the same effect as a head and back fire coming together. Junction zones created by spot fires joining will have a similar effect, yet spread the high heat concentrations around and not in a continuous path as with the other situations mentioned (Sackett 1968).

Season

Burning during different times of the year can be used to take advantage of various phenological and physiological conditions of the trees to modify their susceptibility to fire damage. Spring and summer may be superior to the traditional fall season for thinning with fire (Harrington 1987). We still recommend initial burning in fall. Repeat or rotational burns can be made at other times of the year.

Ambient Conditions

Taking advantage of ambient conditions on any given burn day is another way of manipulating the fire environment. Death of pine needles occurs when temperatures are sustained above 125° F (Hare 1961). When air temperatures are already high, needle and bud temperatures do not have to be raised much by the fire to kill plant tissue. Likewise with low humidities and drier fuels, less energy is required to evaporate moisture and therefore is more available to heat the crowns.

Thinning ponderosa pine stands with prescribed fire is an art that takes skillful manipulation of the fire environment and the fire itself. Conditions are so diverse, spatially and temporally, that the burner must skillfully prescribe the proper treatment for each thinning situation.

NATURAL REGENERATION IN SOUTHWESTERN PONDEROSA PINE FORESTS

Ponderosa pine is considered a difficult species to regenerate in the Southwest primarily because regular periods of moisture stress are caused by droughts and competition from grasses early in the growing season (Larson and Schubert 1969a, Pearson 1950). Numerous papers point out the difficulties encountered with planting, seeding, and natural regeneration (Heidmann et al. 1982, Larson and Schubert 1969b, Rietveld and Heidmann 1974). Prescribed burning is valuable for increasing the probability of obtaining natural regeneration, especially on the silty, volcanic soils of northern Arizona.

Schubert (1974) listed the optimum conditions for obtaining adequate natural regeneration:

1. A large supply of good seed.
2. A well-prepared seedbed.
3. Little or no competition from other vegetation.
4. A low population of seed-eating insects.

5. Sufficient moisture for early seed germination and seedling growth.

6. Protection from browsing animals and insect pests.

Certain of the conditions are unmanageable (precipitation, seed crops), others are partially manageable (seed eaters, insects), and a few can be managed to improve regeneration success (quality seedbeds, competing vegetation). Soil moisture seems to be the most critical factor in seedling establishment. Therefore, any activity that results in an increase in available moisture or an increase in soil volume tapped for moisture by roots would be beneficial. Mineral soil with a light litter covering is generally thought to be the optimum seedbed (Pearson 1950, Schubert 1974), because it allows best seed and seedling contact with available moisture. Much precipitation can be absorbed by a deep forest floor and then lost through evaporation without reaching the root zone (Clary and Ffolliott 1969).

Fire Effects on Natural Regeneration

Removal of forest floor material is beneficial. Pearson (1923) noted long ago that spots where slash piles burned produced large numbers of rapidly growing pine seedlings. Reduction of grass competition was the suggested benefit. Reports by Weaver (1952) and Ffolliott et al. (1977) showed much greater pine seedling establishment on burned than unburned seedbeds. Heidmann et al. (1982) studied sites of best natural ponderosa regeneration in a harvested watershed in central Arizona. Of the sites adequately stocked, 70% had been burned before a moderate cone crop was produced. Harrington and Kelsey (1979) illustrated the deleterious effect of a deep organic layer and competing vegetation on ponderosa establishment in Montana. An additional finding was the much greater size of pine seedling crowns and roots in burned plots, presumably from an increase in available nitrogen.

As part of the fire research at Fort Valley Experimental Forest, burned and unburned seedbeds were surveyed after the 1976 seed crop was produced (Sackett 1984). Burned plots had 2600 seedlings per acre compared with 833 seedlings per acre on unburned controls. After 2 years, no seedlings remained on control plots, whereas burned plots still supported over 500 seedlings per acre. In a companion study, seeds falling on an undisturbed forest floor seldom reached mineral soil (Haase 1981). Sackett (1984) showed a high correlation ($r^2 = 0.85$) between quadrat bare area (square feet) exposed by fire and quadrats stocked: 83% of the new pine seedlings germinated on microsites where the forest floor was partially or totally consumed by fire. Another confirmation of this benefit came from a prescribed burning study in southwestern Colorado (Harrington 1985), where 20 times more pine seedlings per acre were located on burned units than on units with unburned forest floors and Gambel oak.

A more recent seedling survey at Fort Valley Experimental Forest revealed a more pronounced regeneration success. In 1983, seeds were cast at a rate possibly rivaling that of 1918. By summer 1984, the burned plots were carpeted with new seedlings. Plots that had recently burned were surveyed extensively, along with the unburned controls. The burned seedbeds averaged over 90,000 seedlings per acre. The unburned plots had 26,000 seedlings per acre. In fall 1984, two of the three previously burned plots were reburned as part of the burning rotation study. One plot had 4 years of litter accumulation and the other had 8 years accumulation. Four years after burning, the following seedling distribution was found: all seedlings were killed on the plots burned with 8 years of litter, 7,800 seedlings per acre remained on the plots burned with 4 years of litter, 15,000 seedlings per acre remained on the plots burned before seed fall, and only 1,200 seedlings per acre remained on the controls.

Not only does prescribed burning provide for favorable seedbeds for germination, it also enhances the growing environment for survival. Soil moisture on burned sites at Fort Valley Experimental Forest was greater than on unburned sites because moisture can reach soil unimpeded by the forest floor material (Haase 1986, Ryan and Covington 1986). Work with soil thermocouple psychrometers at Fort Valley[3] confirmed Haase's findings. On burned sites, soil moisture was slightly to significantly more available in the 6- to 12-inch soil depths than on unburned sites. Since tap roots of seedlings exhumed on burned sites at Fort Valley are generally 12 inches long after the first full growing season, more moisture apparently reaches the major rooting zones where the forest floor has been consumed.

Soil temperatures are also higher on burned seedbeds. At Fort Valley, Milne (1978) found burned soil averaged 9° F warmer than unburned soils for the time period for active pine seed germination. Since germination of southwestern ponderosa pine seed is temperature-dependent (Larson 1961), burned sites should favor earlier and more rapid seedling emergence. More extensive monitoring of soil temperatures at Fort Valley[4] showed soils to be warmer on burned sites, but rarely high enough to be damaging. Warmer soils could also result in larger-rooted seedlings

[3]Data on file with Michael G. Harrington at the Intermountain Fire Sciences Laboratory, Missoula, MT.

[4]Data on file with Stephen S. Sackett at the Forest Fire Laboratory, Riverside, California.

(Larson 1967), which should have better survival during the normal dry periods.

Favorable seedling development on seedbeds with improved moisture conditions, and warmer soils is also enhanced by greatly improved nitrogen availability. Covington and Sackett (in press) showed a large increase in available nitrogen from initial prescribed fires and rotational burning. The increased nitrogen on burned seedbeds allows seedlings to attain deep roots and large crowns, which facilitate survival through fall drought and the first winter. Since seedlings are generally much stouter on a burned site, perhaps a greater resistance to frost heaving, common on basalt sites in northern Arizona, is developed.

Preparing Seedbeds With Prescribed Fire

The effectiveness of prescribed fire in consuming the forest floor for hazard reduction was discussed earlier in this paper. The same process produces seedbeds of various qualities. Fires that consume the forest floor, leaving little organic matter, create microsites that surpass unburned areas in moisture and nutrient status. The burn prescription parameters listed earlier for fuel reduction burning also apply to burning for seedbed production. Once the weather and surface fuel moisture conditions have been met, the humus moisture will determine the amount of forest floor consumed, and therefore the quality of the seedbed.

Pattern of Consumption

Even though soil with minimal organic covering is optimal for seedling establishment, the entire site need not be burned to that degree. In fact, a burn of that severity would be hazardous to conduct and would likely cause extensive damage to much of the vegetation and soils. Experience shows that a ponderosa pine forest floor burns in uneven patterns, with microsites of mineral soil alternating with unburned islands. This pattern is probably due to variations in fuel moisture, forest floor bulk density, and consumption of woody fuels. So, a burn consuming 50% to 75% of the forest floor would have a variety of partially to almost completely burned microsites. Partial forest floor consumption is preferred, because high quality seedbeds result and other site characteristics generally are not damaged.

Fuel Moisture Contents

It is difficult to propose specific forest floor moisture contents at which an optimum seedbed will result from burning. This is because of the variable pattern of consumption. Generally, in dense, or otherwise fully stocked groups within stands, prescribed burns will create few mineral seedbeds, probably because of the high forest floor bulk density. However, on sites where mature trees have been or will be removed, fires burn to mineral soil within a large range of moisture contents. Places that do not need seedling regeneration probably will not have much mineral soil exposed, and the places where pine regeneration is desired will have mineral soil exposed by fire. Given the weather conditions and surface fuel moistures mentioned earlier as general burning guides, applying fire with humus moisture contents between 25% and 65% will likely result in adequate mineral soil exposure.

Follow-up

When a crop of seedlings establishes, it is important to defer the suggested rotational burning program for a number of years to keep from injuring the young trees. If seedlings are needed on a particular site (understocking), not much fuel will accumulate after the initial burn. We have found seedlings 6 feet high at age 10 that are growing where old-growth pines have died. They have survived successive fires because there was no fuel close by to scorch them. Once the trees can withstand some fire, the interval between burns should be relatively short. Less fuel produces a less intense, uneven burn, which makes it easier for seedlings to survive the fire environment.

Silvicultural Treatments

The use of prescribed fire in conjunction with different silvicultural treatments in the Southwest has not been researched. However, dense slash fuels should be reduced, just as should forest floor fuels, to facilitate pine seedling establishment. One lesson learned years ago was that large clearcuts followed by intense burning failed to promote new generations of ponderosa pine seedlings (Schubert 1974). Small group selection, shelterwood, or seed tree cuts followed by broadcast or pile burning favors seedling establishment and reduces the fuel hazard. Fire has not been effective in reducing grass or shrub competition (Schubert 1974), but research in different seasons of fire application suggest a possible alternative to the current use of mechanical scarification (Harrington 1985). Lessons can be learned from the Pacific Northwest, where fire is used with various ponderosa pine silvicultural treatments for logging and thinning slash disposal, brush reduction, mistletoe control, as well as site preparation (Barrett 1979).

CONCLUSIONS

Very few forest ecosystems compare with southwestern ponderosa pine in the importance of presettlement fire for maintenance of forest health and stability. Fire history from

this region confirms this. Prescribed fire, in mimicking the natural role of fire, can be an ideal tool for accomplishing many forest management objectives.

The successful reduction of natural fire from ponderosa pine stands within the last century has created hazardous, unhealthy forest conditions. With careful fire use under the general guidance of the prescriptions and cautions presented earlier, the vigor and stability of these forests should return. With time, we may even be able to improve upon the multiple-use production of presettlement, natural conditions. The ideas and prescriptions presented here are very general. Prescribed burning anywhere is site-specific. Land managers must learn how prescribed fire relates to their resources, develop a prescription pertinent to each situation, and monitor results. Each fire is a new experience; therefore, the learning process never ends.

The state-of-the-art in using fire for fuel management, overstory thinning, and natural regeneration is only in the beginning stages of development. Other aspects of prescribed fire have not been addressed in this paper or studied. In addition to the tangible parameters, the intuitive nature or art form of fire application is the key to attaining success. Anyone who has the desire to use fire to help manage land resources must learn both the art and the science aspects. Desired results can be achieved only when practitioners master the techniques of burning and learn the principles of fire ecology.

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Social/Political Obstacles and Opportunities in Prescribed Fire Management[1]

Terry C. Daniel[2]

Abstract--Obstacles to public acceptance of prescribed fire include misunderstanding of fire in forest ecosystems, concern about risks to life and property and assumed adverse effects on scenic and recreation values. Increased appreciation of the ecological, safety (fuel reduction) and long-term aesthetic benefits of prescribed fire could help to solidify public support.

Controlled burns, let burns, or prescribed fire in one form or another has become an increasingly important tool in the public land manager's kit. As an agent of environmental change fire has many things to recommend it. However, there are frequently problems in obtaining public support for the purposeful use of fire in the management of public lands. Objections may be based on visibility reduction and smoke pollution in recreation areas and nearby communities, on the real or imagined danger of the fire escaping and destroying private homes and property or on the unsightliness of scorched vegetation and charred ground. In short, fire in the woods is controversial.

The controversy surrounding the use of prescribed fire can be viewed in the context of a more general dilemma that confronts public land managers:

> The public demands an increasing array of goods and services from the National Forests and other lands, but at the same time insists upon protection of natural areas and natural processes.

These demands need not be completely incompatible, but they do require exceptionally creative and conscientious application of multiple-use management strategies, improved understanding of ecological processes and greater use of modern planning and management technologies. There are several potentially serious problems that must be overcome. First, some resource management policies and practices are not sufficiently sensitive to the value that the public places on nature. This can in part be attributed to the slow response time of large agencies to changing public priorities. However, some of the problem arises because, even with the best of intentions, resource managers still have much to learn about complex natural processes. Second, the public is not always well informed about the "natural processes" that they desire to have protected. The public with which the land manager must deal has become increasingly urban, with relatively little direct contact or experience with natural environments or ecological processes. Finally, given the extent of previous human disturbances and interventions into natural ecosystems, including global air pollution, acid precipitation and the "green house effect," there is some question about whether there are any "natural" areas or processes left to be protected.

It is somewhat ironic that prescribed fire should fall victim to this natural resource management dilemma. Fire is after all a "natural process." Indeed, fire is an essential part of many forest ecosystems. Why then should the public object to prescribed burning in forests where fire is a natural agent of environmental change? There are no doubt many reasons for public concern and resistance, as there are many different concerned publics. This paper will review three general "obstacles" to public acceptance of prescribed fire: (1) public misunderstanding of the role of fire in forest ecosystems, (2) the real and perceived risks of prescribed fires and (3) the adverse effects of fire on aesthetic and recreation values.

Obstacles

Understanding Fire Ecology

As noted above, in spite of the high value generally ascribed to nature, the public frequently does not adequately understand ecological processes. In particular, the time and geographic scale of natural processes seems to be difficult to comprehend. Professional land managers have generally been educated to appreciate the natural scales of time and space. Their view of the land is often based on surveys and remote sensed data covering hundreds of thousands of acres. Through formal models and less formal experience, managers

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[2]Professor of Psychology and Renewable Natural Resources at the University of Arizona, Tucson, AZ, 85721.

routinely project and plan for changes in the land that may not occur for decades. In contrast, it is often difficult for the less experienced public to appreciate environmental rewards that are even a few years in the future, much less those not expected for a century or more. Similarly, from the public perspective 100 or 1000 acres of forest damaged by fire can appear to be a catastrophe—especially if those few acres happen to be around a favorite recreation site or in the view from the front porch of the cabin or condo.

A corollary to the public's misunderstanding of natural processes is that the natural role of fire in forest ecosystems is poorly understood. Several studies (e.g., Cortner et al. 1984; McCool and Stankey 1986; Taylor and Daniel 1984; Taylor et al. 1986; Zwolinski et al. 1983) have shown that the public tends to think of forest fires as being much larger and more destructive, especially to wildlife, than normal (natural) fires tend to be. Further, the public attributes a much greater percentage of forest fires to human carelessness or arson than is actually the case.

While there are trends toward greater public sophistication in recent years, there is still considerable confusion, including misunderstandings about the distinctions between prescribed or controlled fires and wildfires.

Part of the public's misconceptions about the role of fire is undoubtedly due to a lack of appropriate environmental education; only recently have a handful of school systems begun to include regular units on ecology and environment in their curriculum. On the other hand, substantial blame for public misunderstanding in this regard must be attributed to highly successful "Smokey-the-Bear" type campaigns. Fire has for the most part been portrayed to the public as an undesirable destructive agent in the forest. The often repeated and very effective theme, "only you can prevent forest fires" has perhaps been taken too much to heart by the public—it may well be applied now to prescribed fires, as well. In any event, it is little wonder that the public has some confusion and mixed emotions about the role of fire in the forest.

Another possible basis for public confusion regarding the resource managers use of prescribed fire may be related to the perceived purpose of the fire. In selling the new philosophy regarding the role of fire in the forest, the historic natural role of fire and its ecological benefits are often cited. Frequently, however, prescribed fires are actually justified by the need to remove "slash" associated with logging or pre-commercial thinning, and for "site preparation," all in the context of commercial timber sales. These justifications may seem appropriate and reasonable to the trained forester (indeed, they may even be required by budgeting policies), but often these purposes are viewed by the public as incompatible with their desire to protect natural areas and natural processes. At the very least, the fact that fire has found economic use in the context of commercial timber sales raises suspicions in some quarters of the public regarding the claimed ecological benefits of fire.

Fire Risks, Real and Perceived

A second obstacle to greater public acceptance of prescribed fire policies is the perceived (and occasionally real) risk that controlled fires will become destructive uncontrolled fires. This is a calculated risk in all prescribed fire situations. The 1988 fires in Yellowstone National Park provided a dramatic illustration of public reaction to prescribed fires that get out of control. A complicating factor at Yellowstone, and in many other forest areas in the West, is that decades of aggressive fire suppression policies have created a very "unnatural" forest. The loss of a natural mix of species and age classes, some experts believe, is one of the primary causes of serious "forest health" problems that seem to be occurring with increased frequency and severity (USDA Forest Service 1988). In addition, the tendency toward extensive even-aged, over crowded and over aged forests has increased susceptibility to the ravages of insects and diseases, which in turn can set the stage for wildfires of unprecedented intensities and scale. In areas where long periods of fire suppression have allowed the accumulation of large volumes of fuels, and fostered changes in species composition so as to produce more vertically distributed "ladder" fuels, the re-introduction of fire as a "natural" environmental change agent is at least problematic.

The risk of an escaped fire is further exacerbated by the increasing "infiltration" of forest areas by human developments; the wildland/urban interface is large and expanding. All indications are that the number of developments extending into essentially wild forest areas will continue to increase. A "natural," undisturbed forest setting is increasingly desired as a site for homes and other developments. Further, "management" of the forest tends to be viewed as in direct conflict with the desire to be near nature and to live in a pristine environment. Thus, there is the opportunity for significant conflict between residents' style of life and aesthetic goals and the forest manager's desire to protect developments from potentially catastrophic wildfires.

Of course, many of these subtleties are soon forgotten when a catastrophic fire actually occurs—human safety and the protection of property, and the manager's responsibility for same, then become the overriding concerns. It is not surprising that forest managers are very cautious about using fire in any situation where human developments or high use areas might be threatened.

Aesthetic and Recreational Effects

Another important obstacle to public acceptance of prescribed fire is the fact that at least the immediate effects of burning are ugly. Large amounts of smoke, whether in the woods where you recreate or in the town where you live, is unpleasant. There are few communities where additional visible particulates in the air are welcomed. Resort and tourism-focussed communities may be particularly sensitive to visible air pollution. Studies have shown that visitors' appreciation of scenic vistas in parks and wilderness areas is adversely affected by reduced atmospheric visibility (e.g., Latimer, et al. 1981; Malm et al. 1981). Such visibility reductions are, of course, a common consequence of prescribed fires. While the effects of smoke are generally of short duration, they are significant in that they can be extensive and affect large numbers of people; smoke is very conspicuous. Further, given increasingly stringent air pollution and visibility degradation regulations, creating smoke may soon be illegal altogether.

The scorch and charring that is produced by the best controlled burn can also have adverse effects on the perceived scenic beauty of forest areas, at least in the short-term. Anderson et al. (1982) studied the changes over time in perceived scenic beauty of a controlled burn in a southwestern ponderosa pine site. Immediately after the burn scenic beauty was substantially reduced relative to pre-fire levels. Over the next 3 to 5 years, however, the perceived scenic beauty of the site improved to recover and then surpass the pre-fire value. By the seventh year after the burn scenic beauty values were beginning to decline back to the pre-fire level. Clearly the immediate loss in scenic beauty was associated with the scorch and charring of the trees and ground cover. As the charring became less evident, the scenic values recovered. Because the burn was successful in removing significant amounts of downed wood (a negative aesthetic feature) and also stimulated new and vibrant ground cover (a positive feature, see Daniel and Boster 1976 and Brown and Daniel 1986), scenic values actually exceeded the pre-fire level after a few years. Finally, as the ground covers matured and new downed wood began to accumulate, scenic values began to return to the pre-fire levels. The effects of prescribed fire on aesthetic values are not simple, and they seem to depend upon when the burned area is encountered.

Taylor and Daniel (1984) studied both the perceived scenic beauty and recreational quality of ponderosa pine forest areas that had been subjected to different intensities of fire at different times from 1 to 5 years in the past. Half of the sites studied had experienced what was classified as "severe" fire, characterized by high intensity fires with substantial tree mortality, as might occur in a wildfire. The other half of the sites had experienced "light" fire, characterized by low intensity burns with little or no tree mortality, as might occur in a controlled or prescribed fire.

All of the sites subjected to severe fire showed very substantial losses in public judgments of both scenic and recreational values, with no indication of recovery over the 5-year period covered by the study. As the Anderson et al. study found, perceived scenic beauty values for the sites representing light fire all improved relative to the unburned control site for the first 3 years (with the first assessment occurring after 1 year). Sites burned 4 and 5 years before the assessment remained at the 3-year level or declined back toward the level for the unburned site. Relative to the scenic beauty results, light fire generally had less effect on judged quality of the sites for recreation uses. Judgments of the quality of the sites specifically for camping, however, showed somewhat greater sensitivity to light fire effects over a longer recovery time than scenic beauty judgments. At the other extreme, judged quality of the burned sites for nature study was relatively insensitive to fire effects. Taylor's paper in this volume presents a more complete description of this study.

Opportunities

Prescribed fires are now widely recognized by professional land managers as potentially very beneficial for forest management and protection. Prescribed fires can have very beneficial effects for wildlife and for maintaining the natural diversity and vigor of the forest. Many of the negative effects of what is now seen as over-aggressive fire suppression policies might be corrected by the careful re-introduction of fire. For example, prescribed fire may be the only economical means for restoring a natural mosaic of species and age-classes over the great expanses of the Rocky Mountain forests. Prescribed fires, while themselves sometimes viewed as dangerous, can in fact be very effective for removing excess fuels and reducing the risk of catastrophic wildfires. The results of the Anderson et al. (1982) and Taylor and Daniel (1984) studies suggest that an appropriate prescribed fire regime could help to sustain, and even improve scenic and recreational values in some forests. Thus, there are substantial incentives for forest managers to find ways to make prescribed fire more acceptable to the public.

If one obstacle to public acceptance of prescribed fire is the lack of adequate understanding of the role of fire in forest ecosystems, it would follow that some form of environmental education would be useful. There is, in fact, some evidence that educational programs or interpretation instruments can be effective in changing public knowledge, and to a lesser extent attitudes toward fire in the forest. For example, Taylor and

Daniel (1984) found that brochures conveying information about fire characteristics and the environmental effects (including ecological benefits) of fires in ponderosa pine forests were effective in changing responses to fire knowledge questions. In that same study, informative brochures were also effective in changing expressed attitudes toward fire; respondents who had read the brochures were more tolerant of low intensity fires and more accepting of the use of fire in forest management. At the same time, however, it is important to note that these same respondents did not change their evaluations of either the scenic beauty or the attractiveness for recreational uses (e.g., camping) of fire-affected forest areas. In effect, the message from those who participated in the education/interpretation program was

> I understand the potential environmental benefits of fire and I am willing to accept some use of fire as a tool for managing forests, but recently burned areas are still ugly and I do not wish to camp there.

In the context of environmental education efforts, a very palatable rationale for prescribed fire could be developed by emphasizing wildlife habitat improvement and the long term protection of valued scenic, recreational or cultural resources. Prescribed fire policies might even be justified by the need to reintroduce fire as a natural agent in some ecosystems, such as parks and wilderness areas. Of course, difficulties may arise due to the apparent conflicts with the messages so well established by previous "Smokey the Bear" campaigns. The earlier fire prevention campaigns were very successful, in part because they were very simple—forest fires are bad and should be prevented—and very dramatic—burned and smoldering forests look terrible and the thought of little animals being burned up alive is very disturbing. In contrast, the environmental education effort needed now is much more complex—some forest fires, under appropriate conditions, can, in time, be beneficial for some types of forests. Instead of the dramatics of the conflagration and smoking charred aftermath, the ecological benefits of fire are much more subtle and require an appreciation of the time scale and dynamics of the forest ecosystem. At the same time, the potential benefits of forest fires must not be over sold. It would be unwise to have the public forget Smokey the Bear's message altogether; the object is certainly not to have people become more careless with fire in the woods.

While the case can be made that fire is a natural and beneficial part of many forest ecosystems, the purposeful use of fire by public land managers is still problematic. Previous fire management policies have frequently created conditions that virtually preclude an immediate return to "natural" (unmanaged) fire regimes. The continued expansion of the "wildland/urban interface," adds to the difficulties by exposing more people and property to risk should a controlled fire get out of hand. As a result, managers can be expected to be increasingly reluctant to use prescribed fire as a tool near these areas. Ironically, this reluctance to risk prescribed fires in these sensitive areas can only lead to increased fuels, and thus increase the risk of a really catastrophic wildfire at some point in the future. This is a typical "pay now or pay later" situation. Some public land managers may choose the "later" option in hopes that later will occur on someone else's watch. Responsible management, however, requires an attempt to communicate the relative short and long term risks, and to meaningfully engage the potentially affected publics in the decision making process.

Like fire risks, aesthetic and recreational effects of prescribed fire depend upon the time at which they are assessed. The general indication (for southwestern ponderosa pine forests, at least) is that while both scenic and recreational values may be negatively impacted in the initial period after a prescribed burn, these value recover rather quickly and can even exceed the pre-burn values within a few years. Coupled with known ecological benefits and the historic "natural" role of fire in some forest ecosystems, these aesthetic and recreational effects could provide a strong basis for public support of prescribed (controlled) fire programs. Smoke, while a relatively short-term nuisance, will probably continue to be a deterrent to prescribed fire programs, especially given the continuing concerns and regulations regarding air quality and visibility degradation. Perhaps the best approach here is to point out that accumulating fuels in forest environments ultimately will burn. Our choice is to accept the inevitable smoke in small manageable installments, or take it in a lump-sum payment at some unpredictable time in the future.

Conclusions

There continue to be significant obstacles to more widespread use of fire as a tool for forest management. While there has been considerable improvement in recent years, there is still substantial misunderstanding in the public of the role of fire in forest ecosystems. Risks to human lives and property, as well as damage to the forest and the wildlife therein continue to be of concern to the public, and this concern can become very potent when a prescribed fire does in fact get out of control. There is also concern about the adverse impacts of fire on forest recreation and scenic values, either as a result of smoke "pollution" or of fire effects on the vegetation. The tension between air quality regulations and the use of prescribed fire in forest management

will no doubt increase in the next few years. Still there are compelling and sound reasons for expanding the use of fire in the management of many forest ecosystems. Uses range from fuel reduction as an effective means of decreasing the risk of future catastrophic wildfires to regeneration and other vegetative change objectives in the interest of timber production or wildlife habitat improvement. Another important goal for prescribed fire is the re-establishment of the natural role of fire in forest ecosystems, including the restoration of a more natural mosaic of species and age classes, and maintaining vigor and resilience of the forest (i.e., creating a "healthier" forest).

Restoring a more natural fire regime to forests is a commendable and appropriate resource management goal, and it is in many ways consistent with public desires to restore and protect natural processes in public forests. Also, evidence has been increasing that, in at least some forest types, appropriate use of prescribed/controlled fire can enhance scenic and recreational values. More often, however, the explicit rationale provided for prescribed fires revolves around timber management objectives (such as the aforementioned "slash disposal" and " site preparation") and the need to protect timber resources. The emphasis on timber management objectives reflects long-standing and well established policies that may have some merit. However, whatever arguments one might wish to muster in support of the importance of timber management objectives, the fact is that there will be little public sentiment for making what is perceived as long-term environmental sacrifices in the interest of economic timber management objectives.

The opportunity clearly does exist to increase public support for the use of prescribed fire on the basis of the ecological, scenic and recreational benefits that could be achieved. It goes without saying, of course, that if prescribed fire is going to be justified on the basis of wildlife, aesthetic, recreational and ecological benefits, it is essential that the use of fire in fact be carefully designed to serve those purposes.

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Smoke Management: An Emerging Profession[1]

Peter Lahm,[2] Douglas Fox,[3] Al Riebau[4]

Abstract.—Smoke management is increasing as an activity conducted by federal land management agencies throughout the United States. Efforts to meet air quality regulations affecting smoke management will succeed as fire managers plan for smoke management as an integral aspect of burn prescriptions. Operational tools are now available for use, though continuing improvement of smoke management hinges on research and development to advance the capabilities of the science. An example of smoke management planning using available tools is presented.

The Southwest has beautiful desert landscapes, forested mountains, and a rapidly growing population. People expect clean air and unrestricted vistas. They complain when their visibility is reduced. Visibility reduction is caused by pollution transported into the region from the large urban complex of Los Angeles as well as from the growing metropolitan area within the region. People are generally not aware that smoke from forest burning has historically been a part of the landscape and their tolerance for smoke is rather low. Diurnal wind patterns exist in the Southwest because of the strong radiative heating in daytime and cooling at night. These winds transport residual smoke from forest burning in the mountains into valley bottoms where towns and cities are often located.

Land management in the region must use fire to accomplish a variety of goals ranging from habitat improvement to natural fuel reduction. For example, the various land management agencies have prescribed fire programs covering 100,000 acres of ponderosa pine fuels and 5,000 acres of chaparral fuels per year. Both pile-burning and broadcast-burning are used to reduce these fuels.

In order to ensure that smoke from prescribed burning does not become a problem, the state air quality divisions in Arizona and New Mexico use a permit system. In Arizona a permit application is made yearly and permission is granted to burn on a day-to-day basis dependent on dispersion and weather conditions. This system generally allows adequate flexibility for both prescribed burning programs and smoke management. But, smoke management must be a significant component of the program.

A Smokey Day in Sedona

The need for smoke management can be demonstrated by a recent incident in Sedona, Arizona. In September 1988, about 400 acres of hand-piled ponderosa pine logging slash (fuel loading was approximately 18 tons/acre) was burned. The burn was on the Mogollon Rim about 2000 feet above and 12 miles north by northwest of Sedona. Knowing that down-canyon nighttime airflows transport smoke into town, the burn boss followed a smoke management plan limiting ignition to only 200 acres each day. Still, Sedona suffered an impairment of visibility on three successive mornings after the burn was ignited. Elevated levels of particulates disturbed those suffering from asthma and allergies.

Factors that contributed to the smoke problem in Sedona are the nighttime drainage wind flow, development of an inversion that limited dispersion, and a large total loading of smoke from a number of sources including fireplaces and wood stoves. At the same time, transport of pollution from long distances into the area may have made a significant contribution.

Before the prescribed burning season, a press release to area media and letters to key people announced the potential for smoke to limit visibility in populated areas. Nevertheless, there was significant public concern about the amount of smoke.

Several points can be made about this incident. Even though the burn boss used a responsible fire prescription and smoke management plan, the outcome was unexpected. Covering piles and burning them when good dispersion is assured, coordinating the allowable number of actively burning acres near sensitive

[1]Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).

[2]Smoke Management Staff, USDA Forest Service, Coconino National Forest, Flagstaff, Ariz.

[3]Chief Meteorologist, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colo.

[4]Air Quality Scientist, USDI Bureau of Land Management, Wyoming State Office, Cheyenne, Wyo.

areas, expanding the burning season, reducing the total acres ignited per day, mopping up, and broadcast-burning rather than pile-burning are examples of options that can be considered. Because of the complexity of both topographic and meteorological conditions surrounding such burns, site burn prescriptions may need to utilize more expensive and complex techniques, including meteorological measurement and modeling technology. In this paper we will apply state-of-the-science planning tools to this example to look at some of these alternatives.

The Professional Smoke Manager

The art of "smoke management" is developing into a profession. To many, the term focuses on regulatory concerns for air quality standards as enforced by federal, state, or local authorities. Regulatory agencies have no responsibility to consider smoke from the ecological perspective understood by a land manager. They generally have no knowledge of the role fire has played historically and needs to continue to play in the management of natural resources. They do not recognize that fuel accumulation must be managed to prevent excessive buildups that lead to catastrophic fires, a predictable consequence of a management policy of no fuel treatment.

Rather, regulatory agencies view burning from an engineering and economic perspective. Burning activities to reduce fuel loads in slash or to prepare planting sites are often viewed as commercial activities in the same sense as a coal-fired power-plant is a commercial activity. Regulators consider smoke from prescribed fires particulate pollution. Their job is to maintain air quality within federal and state standards.

While land managers understand that fire has an historical presence and that a little smoke from prescribed fires is better than a lot of smoke from wildfires, regulators worry about each day's pollution loading in the airshed. Their job is to manage these loadings. And in the conduct of this management, the largest sources are considered for control first. If calculations and data suggest that forest burning is the primary source of small inhalable particulates (e.g., PM_{10} particles less than 10 m in diameter), then forest burning becomes a regulator's target. Thus, it is not surprising that land managers and regulators approach smoke management with somewhat different perspectives.

Regulations affecting prescribed burning fall into three categories. Ambient air quality standards are set at a specific concentration selected to ensure that public health will be protected. The Federal Clean Air Act of 1977 mandates these standards. For example, the ambient standard for PM_{10} is 150 g/m^3, a 24-hour average concentration that is not to be exceeded more than once a year. This standard should not be violated on areas "off the site" of a fire.[5]

The second type of standard results from the Clean Air Act Prevention of Significant Deterioration (PSD) regulations. These can be described as esthetic standards. They are formally applied to stationary sources (e.g., coal-fired power plants) and allow for incremental degradation of air quality by specific pollutant increments above a baseline set by the first applicable industrial development.

These regulations also require the protection of air quality related values, including visibility, in Class I areas.[6] While it is clear that Congress wished Class I areas to be protected from visibility degradation caused by industrial sources, there is no mention of prescribed fire related visibility reduction in the PSD section.

The Clean Air Act mandates a state-centered regulatory process. Each state has the responsibility to establish whatever regulatory structure it wishes, subject only to the constraint that its ambient standards are at least as stringent as the federal standards. Furthermore, State Implementation Plans (SIP) codify a process to achieve air quality goals. A state may, therefore, choose to regulate smoke from prescribed fire.

The third type of regulation used to control fire emissions is requiring state and local permits for open burning. Although there are often provisions allowing exemption for agricultural burning, slash burning or range treatment by burning usually are not considered agriculture. Open burning requirements often include provisions that the burner not impugn the health, safety, well-being, or enjoyment of the public. Specific safety provisions to insure that highways or towns are not filled with smoke are common and are enforced.

If smoke management is to develop as a profession, it must develop techniques to allow prescribed burning to be conducted without violating air quality regulations. The professional smoke manager must understand such things as air quality regulation developments and implementation, the fundamentals of fire behavior, the use of fire as a land management tool, the effects of fire emissions on health and welfare, the influences of meteorology on dispersion/transport of fire emissions, the basics of simulation of the dispersion of smoke, and the practice of public information. The mix of skills needed by smoke managers is not, to our knowledge, currently supplied in any university program. It is one which

[5]For air quality impacts, "off the site" normally refers to the boundary of private property within which the public does not have general access. For prescribed burns, the Wyoming Air Quality Division has arbitrarily defined "off the site" as 1 km from the advancing fire front.

[6]National parks, wilderness areas, and similar lands above certain size categories that existed in August 1977 when the Act was passed are set aside for special air quality protection and minimal air quality degradation.

we feel must be recognized and encouraged for courses of study at universities.

Current Practices in Smoke Management

In some states elaborate smoke management plans and procedures have been developed. Oregon fires are scheduled based on centralized daily determinations of dispersion meteorology. Dispersion forecasts are made for seasons when fire is allowable (during those times when transport of smoke into Class I areas will not likely lessen the enjoyment of visitors). In Wyoming fires are permitted by the state air quality authority only after they have been modeled to demonstrate that air standards are not likely to be violated. In most of the Southwestern states, open burning regulations of some kind already exist.

Operational tools applicable to smoke management now include monitoring devices, databases, and models. Monitoring devices include particulate monitors for PM_{10}, meteorology measuring devices such as remote automatic weather stations (RAWS), and various "sounders" that collect information on wind, temperature, and humidity distribution at various atmospheric levels.

Examples of databases which can be used are the BLM Initial Attack Management System (IAMS) data and the developing USFS Weather Information Management System (WIMS). Models available for use are the USFS Pacific Northwest Station Emission Production Model (EPM) for calculation of fire emissions and the USFS Rocky Mountain Station/ BLM Topographic Air Pollution Analysis System (TAPAS, an interactive system of terrain-based dispersion and wind simulation models) (Fox et al. 1987).

All of these tools require knowledgeable personnel to use them. None of them is so simple that it can be used without thought. As an example, consider the collection and use of weather data.

Collection and Use of Weather Data

Often the most significant smoke management question is, What is the best way to use weather data? Classically, fire weather data has been collected as input to the National Fire Danger Rating System (NFDRS) (Deeming et al. 1972). These observations have been archived and are available through the National Fire Weather Data Library (NFWDL, Furman and Brink 1975). NFDRS provides indication of fire potential and NFWDL provides an historical fire weather data base. But these systems include only a daily observation at the location of the weather station, often a valley bottom near a Ranger Station. They sometimes do not represent the actual weather condition at the burn site. Currently efforts are underway in the USFS to develop a Forest Service Weather Information System (WIMS, Bass et al. 1988). WIMS will be a comprehensive microprocessor-based, graphics-oriented system that will integrate data and information from the following:

- RAWS—the existing network of fixed Remote Automated Weather Stations.

- P-RAWS—portable RAWS deployed at an activity site for a limited time period.

- NFDRS—the on-line weather and interpreted fire parameter national output.

- BLM-IAMS—the BLM Initial Attack Management System of lightning and weather information.

- NOAA/NESDIS/NWS—weather from remotely sensed data on surface temperatures, cloud cover, temperatures, and soil moisture; coupled with numerical models to forecast weather and biomass moisture.

- Other electronically available data and model products.

By the mid-1990s WIMS will be integrated with Geographic Information Systems (GIS) to provide data interactively. In the interim these data are not available to the on-the-ground smoke manager. RAWS and AFFIRMS (NFDRS) data are available, but of limited utility to the smoke manager because of sparse collection density and the unavailability of upper air data at the fire weather site.[7]

Complete upper air data would provide the smoke manager with an indication of the wind and atmospheric stability in the vicinity of a burn. Coupling this data with a detailed flow model for complex terrain, such as TAPAS, the manager could then simulate where smoke would go and calculate how much visibility would be reduced and particulate concentrations increased when it arrives. Unfortunately, upper air patterns do not remain fixed or stationary. As the sun goes down,

[7]The best measurement device to determine upper air wind patterns is the balloon. Two separate types of balloons with associated instrumentation are useful. One is a tethered balloon, about 3 m long, which lifts an instrument package. The instrumentation sends back data on the temperature (wet and dry bulb), wind speed, wind direction, and pressure. These data allow the plotting of wind with height up to approximately 500 m. The instrumentation is reasonably portable so that data can be obtained at a number of locations. A second balloon technology uses free-flying balloons about 1 m in diameter that have small attached instrumentation packages. These balloons fly free and rise through the atmosphere. By tracking them with theodolites, a picture of the wind speed and direction can be obtained. The instrument package sends data on temperature, pressure, and humidity back to the ground. This package provides a complete description of the atmosphere along the balloon path.

the surface cools and drainage flows develop, and the atmospheric boundary layer collapses and traps smoke underneath it. These phenomena are predictable in general, but not in specific or particular. Thus, even with the relatively sophisticated tools described, smoke will often end up where it is not desired—particularly while a fire remains in a smoldering stage at night.

While the future of smoke management lies in the application of measurement technologies along with modeling, we would like to illustrate the use of a simple smoke dispersion screening model, the Simple Approach Smoke Estimation Model (SASEM) (Riebau et al. 1988), in a real smoke management planning situation.[8] Although the more complex models in TAPAS can provide a better prediction, particularly when coupled with good on-site data of smoke trajectories, the basic features of smoke management can be illustrated with SASEM.

We suggest that smoke managers should use SASEM now as a tool for planning before burning. Figure 1 illustrates three levels of modeling progressing from simple screening to complex research models. As a screening model, SASEM will provide estimations of visibility and particulate concentrations inexpensively with a margin of safety.

Arizona Broadcast-Burn Example

Let's consider an example where, in the fall, a forest manager plans to broadcast-burn an area of approximately 3,500 acres to reduce/remove decaying cull logs, natural downfall, and remaining debris from commercial timber harvest. The fuels involved average 19 tons per acre: fire specialists on the forest choose the following fire weather parameters:

Fuel type	Fuel moisture *(%)*	Fuel weight *(tons/acre)*
Live fuels	n/a	n/a
1-hr fuels	5 - 15	00.2
10-hr fuels	6 - 15	00.8
100-hr fuels	7 - 18	04.3
1000-hr fuels	n/a	13.8

Air temperature *(°F)*	Relative humidity *(%)*	Wind speed *(mph)*
50-80 (day)	15 - 50	1 - 6
30-60 (night)	15 - 50	1 - 6

So that scorch height would not exceed 13 to 15 feet to avoid damaging the standing trees (pole height), it would be unmanageable to burn all 3,500 acres in one session, so all burns are limited to 200 acres to complete the flaming phase of combustion within 10 hours.[9] Under the conditions of the prescription, fire line intensities are calculated to be from 41 to 123 BTU/ft/sec.[10]

SASEM was designed to model smoke emission and dispersion from just such fire prescription information. For each fire SASEM requires the following input data:

- number of acres to be burned
- fuel loading in tons per acre
- fuel type (in this instance woody was used)
- fire line intensity

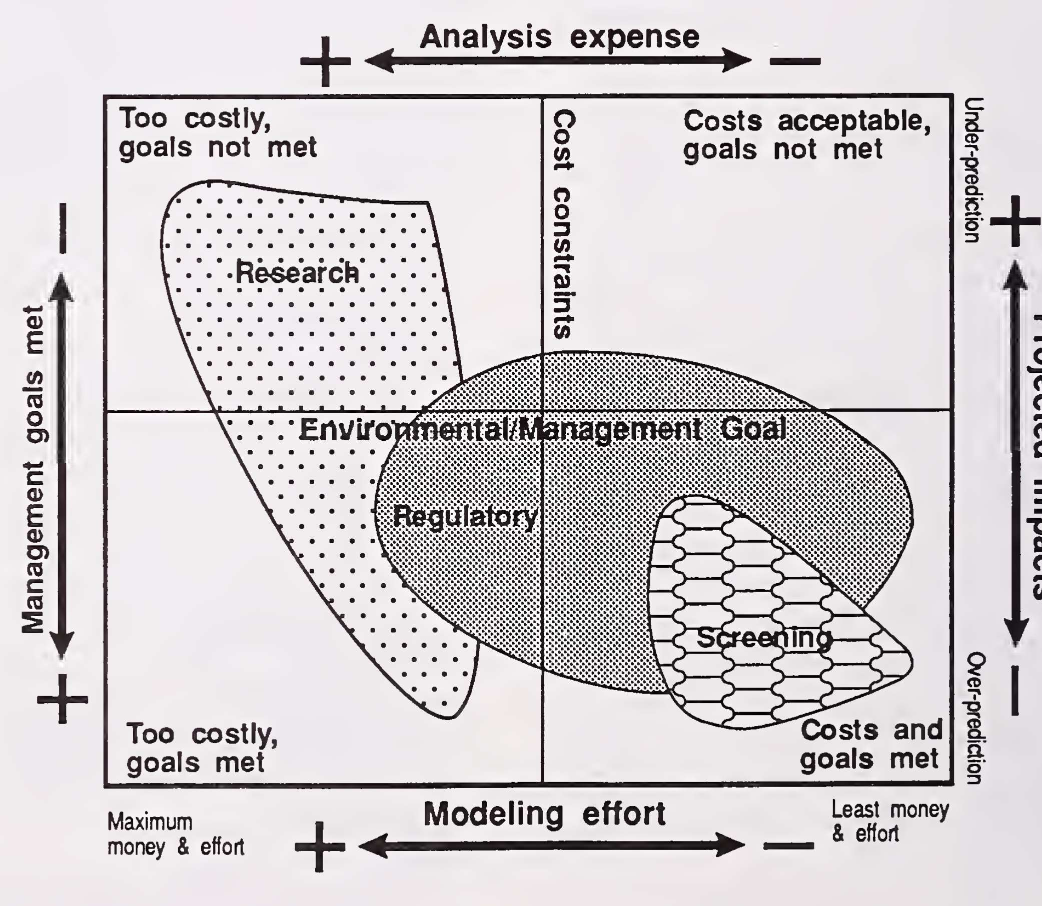

Figure 1.—Models: research, regulatory, screening (smoke management purposes).

[8]Screening models are simplified models that are deliberately designed to over-predict impacts. By over-prediction of impacts, screening models provide a quick estimation of the worst case possible; if such a model were to provide exact estimations of monitoring data, it would be a failure as a screening model. By predicting the worst possible impacts, SASEM provides managers with a wide margin of safety. Thus, if SASEM predicts that smoke management objectives will be met (i.e., visibility impairment will be minimal and air quality standards will not be violated), there is no need for more complex analyses and the project can go forward. If SASEM predicts undesirable impacts, two courses of action are possible. One is to reduce emissions; the second is to utilize a more accurate, less conservative model.

[9]It was estimated that smoldering might go on for up to 3 days.

[10]Roy Hall, Fuels Management Technician, Coconino National Forest, personal communication.

- burn duration in hours
- wind speed in MPH.

As an indication of the maximum loading potential, SASEM, on the entire 3,500 acres to be burned, predicts 381 tons of particulate to be produced (11.4 g/Kg of fuel). In this case, SASEM predicts that the maximum offsite concentration would be 9 times the ambient standard (approximately 1,400 g/m^3) under poor dispersion conditions (e.g., PG Class E or F) and 1 mph windspeed. The burning of 200 acres at a time results in offsite concentrations that would not violate ambient standards except in limited areas under the poorest dispersion conditions and low wind speeds.

One of the useful features of SASEM is its ability to estimate visibility (atmospheric optical clarity) at remote receptors. For this example, we have located a receptor 11 miles from the fire. SASEM reports visibility in miles of visual range. In simple terms, visual range indicates how far one might be able to see under the worst case if smoke were transported directly to the receptor (e.g., plume centerline). To demonstrate the usefulness of this tool for fire planning, SASEM was run with several burn durations to show the different impacts possible.[11] Figure 2 presents these estimates for different meteorological conditions. Varying burn duration and thus emission rate from the fire changes the visibility impact predictions.

[11]It should be noted that varying the burn duration could vary to scorch height and thus not meet the fire objectives. Visibility objectives must then be considered in the matrix of other burn objectives.

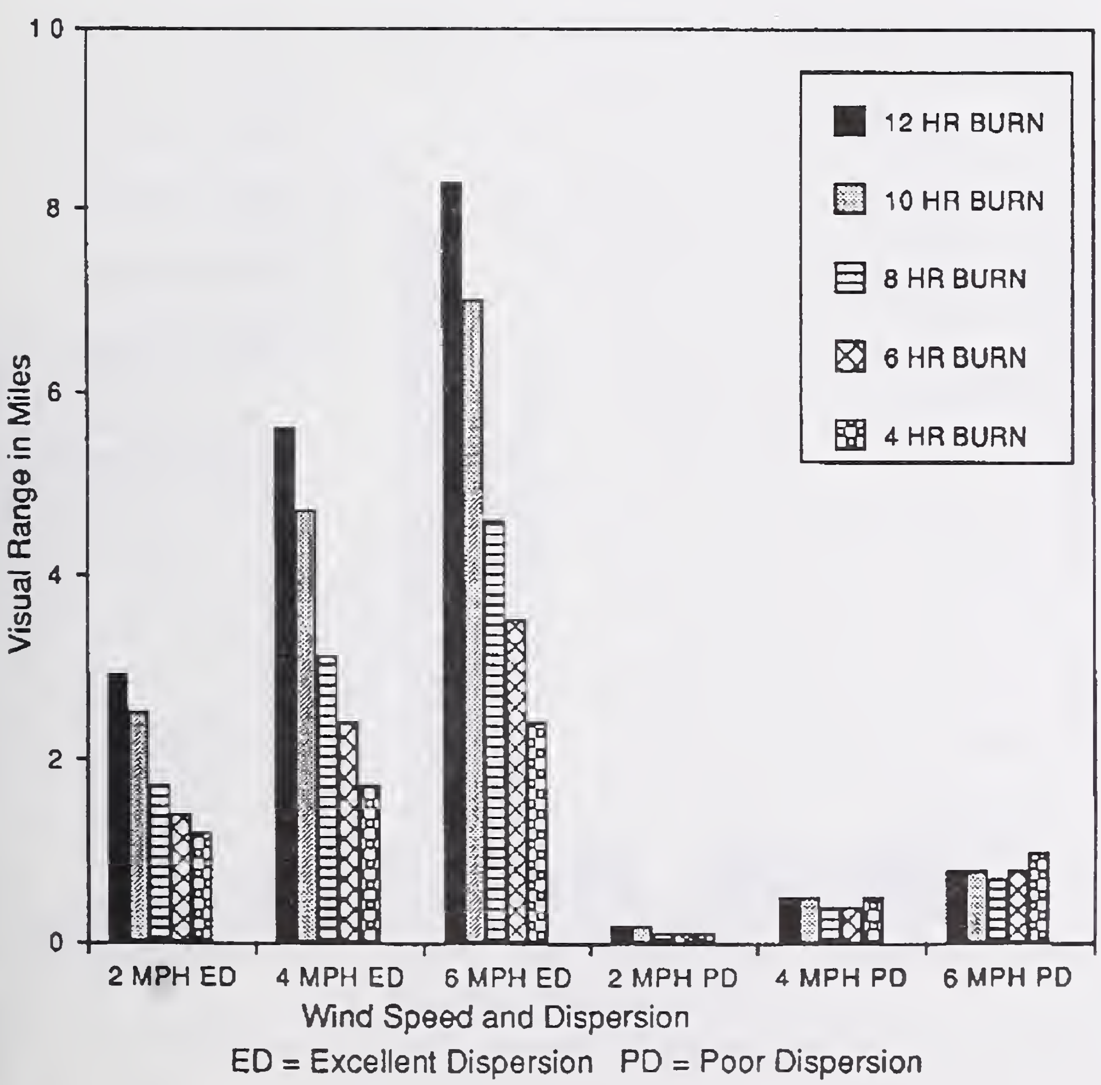

Figure 2.—Visual range with burn duration, Sedona, Arizona (200 acres, woody fuels).

As can be seen from the example SASEM analysis as depicted in figures 3 and 4, modifying the fire's management will result in little lessening of impacts to visibility. Figures 2 and 3 clearly show that the fuels involved will produce enough smoke to cause visibility impairment if the smoke gets there. Modifying either the acreage or the duration of the burn would not significantly lessen impacts. If this planned burn is in a similar location to our example 11 miles NNW of Sedona, Arizona, it is clear that under low wind speed and limited dispersion conditions typical of that routinely occurring in drainage flows, SASEM predicts visibility could be reduced to less than 1 mile and particulate loads could exceed 4-6 times the ambient standard. As SASEM is designed as a screening model, these high pollution predictions are worst case. They should never actually occur because if they did, the model would not be overpredicting. However, the SASEM prediction does "red flag" this burn for managers. Although the actual predictions are larger than what is likely to actually happen, the manager needs to consider that there may be other burning in the region and weigh the risk of polluting Sedona against the likely resultant consequences. SASEM illustrates that the "risk adverse" choice for the fire manager is to extinguish the fire before drainage flows transport the smoke to Sedona. If this alternative is unacceptable, the manager will need to develop more information by investing in meteorological measurements and using the more expensive, complex, and accurate models contained in TAPAS.

The Southwestern Smoke Management Perspective

The Southwest has several air quality issues before it. The Sedona incident underscores the need for a professional commitment to smoke

management by the groups that conduct prescribed burning. Compliance with the provisions of state programs, and professional smoke management will increase the costs of prescribed burns. The possibility of periodically having to fully suppress a prescribed burn to alleviate smoke impacts to a sensitive area is real. Acceptance of appropriate costs by agencies is necessary to integrate effective smoke management as a component of prescribed burning.

Conclusions

The profession of smoke management will be defined by smoke management professionals themselves. Paradoxically, this profession is still so undefined that few, if any, true smoke managers exist. The archetype smoke manager must be trained in fire behavior and ecology, fire management, dispersion meteorology, fire emissions calculation, air quality regulations and regulatory processes, dispersion and ecological modeling, and public relations. This particular and challenging mix of skills has not yet been formally addressed by the nation's universities, although various federal land management agencies have attempted to address training in these areas through internal programs.

Smoke management means that the effects of smoke are incorporated into burn plans before burning, monitored during the burn, and assessed for impacts after the fact. We suggest that SASEM can be a valuable screening tool and should be used in the development of burn plans with potential to impact sensitive areas.

Smoke managers also need a mix of tools to practice their profession. The most important tool needed by smoke managers is adequate meteorological data for both surface and upper air conditions. Without these data smoke management can not be practiced in any real sense. Another tool is accurate emission factors for open fires. The necessity of having accurate and dependable factors is paramount. Also, an important tool is a set of dispersion models in addition to SASEM which can accurately predict smoke transport and pollutant concentrations at sites remote from the fire itself.

To develop these skills and tools there is a need for continued research. Emission factors must be developed that are specific for fuel types common to the Southwest. Research has been done to develop emission factors for fuels in both the northwestern and southeastern United States. These factors cannot be applied to southwestern fuel types without field testing to see if they are applicable. In addition, the collection and dissemination of local meteorological data to smoke managers is needed. Basic research on the influence of local meteorology on smoke transport and dispersion will aid the smoke manager as well as the fire planner. Screening models like SASEM err on the side of overprediction. They bias uncertainties to cause overprediction. However, overpredicting the consequences of prescribed burning can be costly. Added attention to the burn, reduced allowable fuel loadings, and limited burning acreages all add to the cost and limit opportunities to achieve management goals.

Modeling of smoke dispersion in complex terrain is an area that needs development. A basic question, such as how high the plume from a fire will rise, must be studied. Much work remains to provide truly reliable modeling tools that are palatable to regulators and more useful to smoke management planning. Research for all of these issues and others we have not highlighted in this paper, when complete must be transferred with their proper background

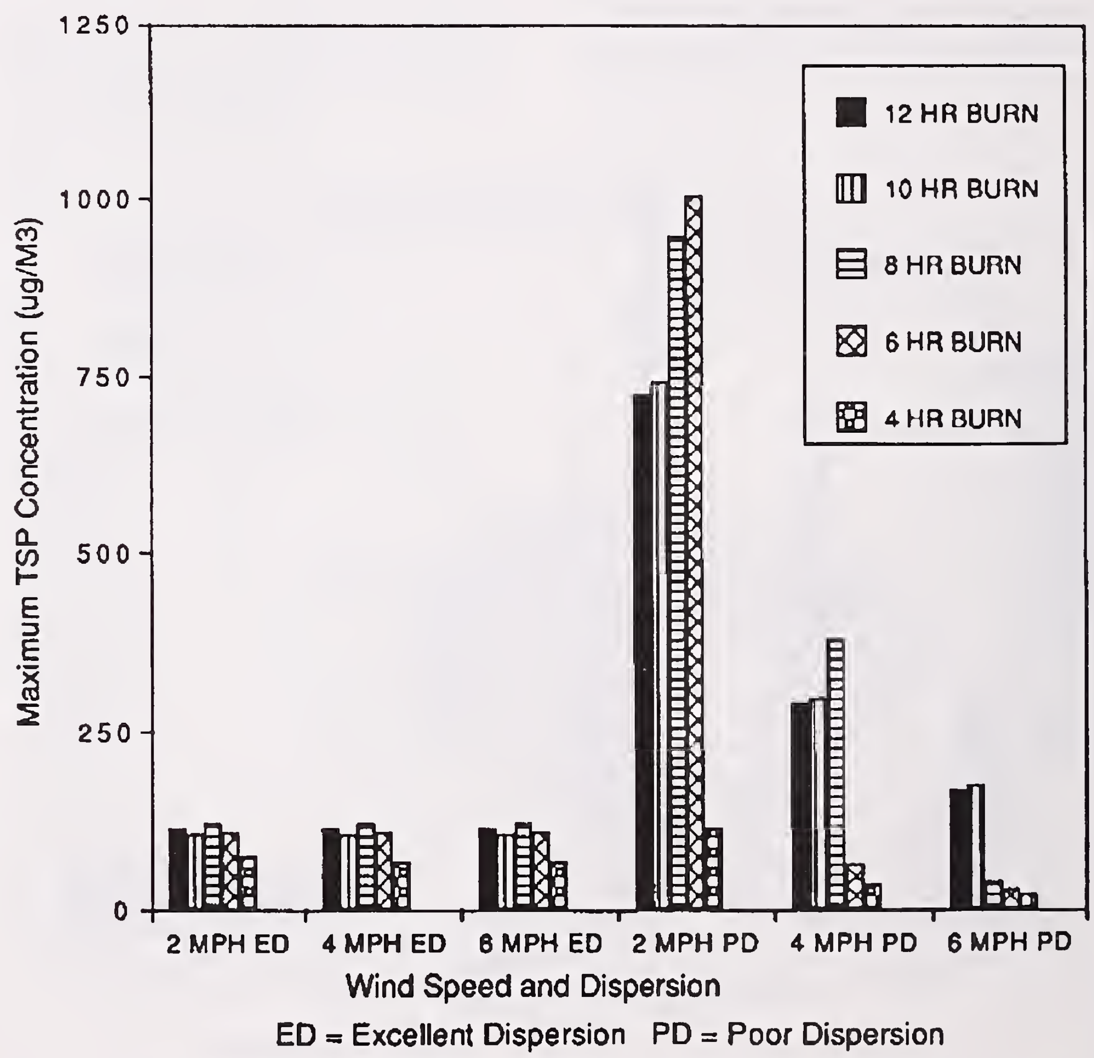

Figure 3.—Max TSP conc with burn duration, Sedona, Arizona (200 acres, woody fuels).

materials into a formal smoke management educational program most properly conducted at universities.

Where will we find professional smoke management in the final analysis? Professional smoke management must be part of every fire plan in a meaningful and useful way. If we don't meet the prescribed smoke management conditions, we don't burn. Professional smoke management must also be recognized in state regulatory programs. Professional smoke management must have a recognized body of professional-level tools backed by solid research. Finally, it must have practitioners who function as professionals.

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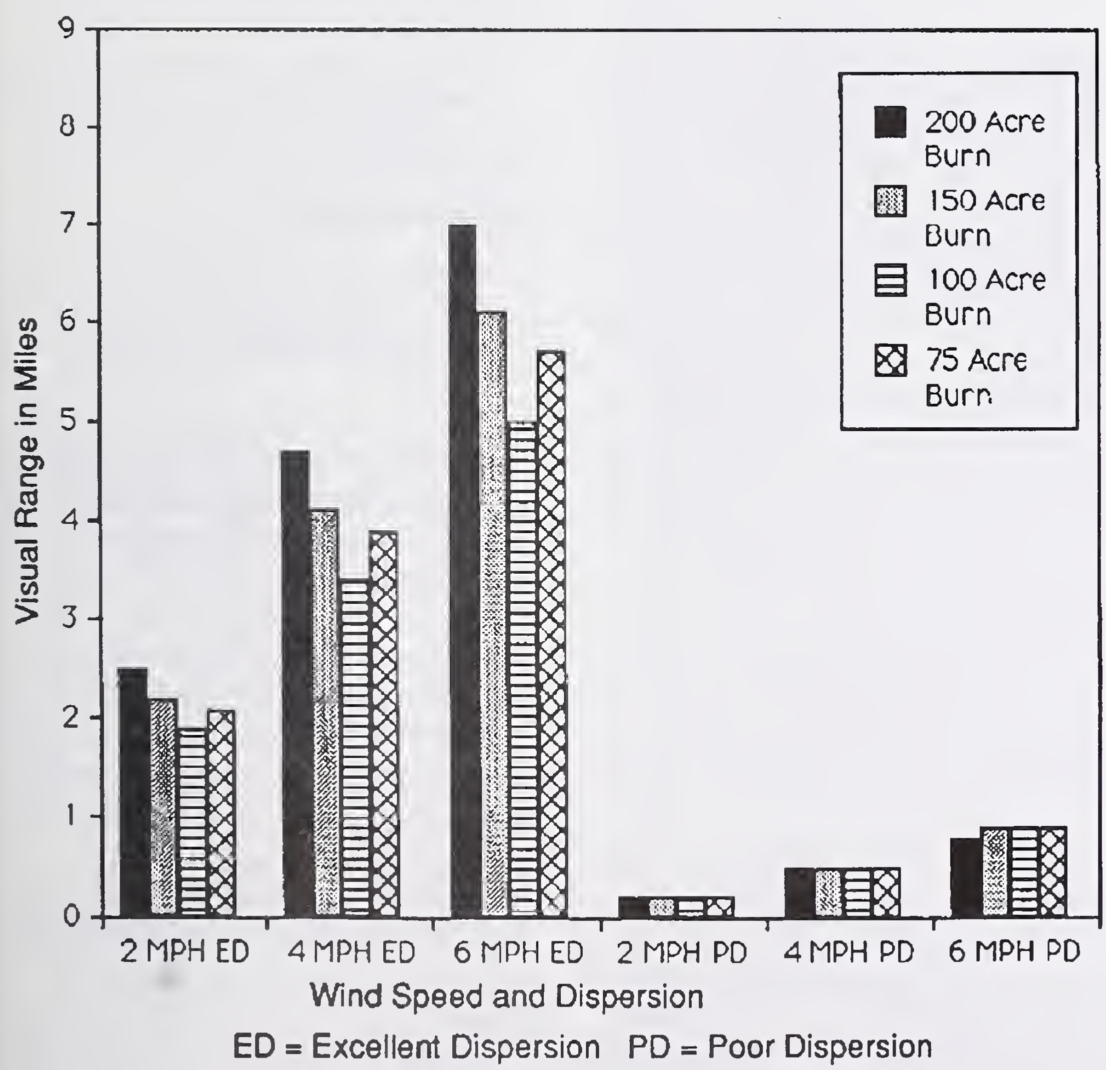

Figure 4.—Visual ranges with burn acreage, Sedona, Arizona (broadcast - woody fuels).

245

Prescribed Fire Monitoring and Evaluation Activities[1]

P.N. Omi[2]

Abstract.—Monitoring and evaluation activities are essential for improving the quality of prescribed burning efforts. This paper reinforces the importance of monitoring and evaluation activities, reviews a case study of current prescribed fire reporting practices, provides a framework for conducting these activities, and suggests refinements.

The effects of prescribed fire activities are not restricted to one point in time. Many fire effects may not be evident immediately after a fire. Traditionally, little emphasis has been placed on monitoring fire behavior for correlation to fire effects, or to evaluating the quality of prescribed burning efforts. To date, greater energies have been devoted to fulfilling burn targets without allowing escapes. To maintain the vitality of your efforts, you will want to monitor and evaluate your progress as you go, so that you can learn from past experiences and update your expectations for the future.

My objectives are: (1) to illustrate the importance of monitoring and evaluation activities in prescribed fire management; (2) to review a case study of current prescribed fire reporting practices and discuss suggested improvements; and (3) to provide frameworks for carrying out monitoring and evaluation activities.

Definition of Terms

Monitoring is the systematic process of *collecting and recording* information to provide a basis for adjusting and improving future efforts. **Evaluation** is a process for appraising the results of a decision, through both qualitative and quantitative monitoring activities. Thus, monitoring is the process of collecting data; evaluation is where you analyze the data to assess the quality of your efforts.

Importance of Monitoring and Evaluation Activities

Monitoring and evaluation activities are important in order to know whether you are achieving goals and objectives as stated in the fire management plan. These activities also provide benchmarks for implementing future improvements. For example, if a prescribed burn doesn't achieve your mangement objectives, then you will want to examine the reasons so that you can improve your future efforts. I strongly believe in the importance of monitoring and evaluation activities as personal education devices, which improve the professionalism of your efforts. Finally, if everyone documented their efforts and pooled information in a shared data base, then we could learn from others' experiences, and perhaps, avoid costly repetition of mistakes.

Over time, managers have developed considerable expertise in the skillful application of fire. Accomplished practitioners and their experiences represent a potentially valuable information source for planning future efforts. Unfortunately, knowledge is not transmitted very effectively by word of mouth. As a result, much useful information about application of fire as a treatment tool remains undocumented and unavailable. Too often, monitoring and evaluation are given low priority, as compared to all the other tasks that must be completed in planning and implementing a prescribed fire. Emphasis on completion of "targets" may compromise quality control efforts. In fact, monitoring and evaluation efforts may only be given lip service in some management plans.

Timeframes for data collection are also important. The timing and reasons for data collection can be categorized as follows:

Preburn: description of conditions prior to burning (ecological, environmental, fuel descriptors).

During burn: to see if the fire stays in prescription; to provide a data base for relating fire behavior to effects.

Postburn: to document effects of fire and conditions subsequent to burning.

We can distinguish two types of monitoring and evaluation efforts in prescribed fire. *Research* involves controlled experimentation, and frequent data collection to discover new facts. *Operational* monitoring and evaluation efforts are conducted to see if a desired outcome is achieved and to record conditions responsible for the outcome.

[1]*Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Professor, Forest Fire Science, Department of Forest and Wood Sciences, Colorado State University, Fort Collins, CO 80523.*

Reasons for Monitoring and Evaluation

There are many reasons for monitoring and evaluating prescribed burns. Some of the more important considerations are given in the following tabulation. These span both short- and long-term concerns, as well as implementation and planning needs.

To determine if a fire remains in prescription

To assess whether objectives are being achieved

To document observed fire behavior for comparison with predictions

To assess immediate and long-term fire effects

To refine and adjust prescriptions

To improve economic efficiency

To apprise line management of progress

To document successes and failures

Although we are focusing on prescribed fire in this conference, many of my remarks can be extended to a variety of fire management activities. These include decisions associated with wildfire suppression, natural fire management, fuel treatment, or determining the appropriate mix of presuppression/suppression forces to employ in your management unit.

Elements of Operational Monitoring and Evaluation Efforts

The overall goal of monitoring and evaluation efforts is to collect enough information to enable evaluation of the degree of success (in achieving objectives). To do so, requires careful planning and consideration of resource management objectives, treatment objectives, and variables to be measured. Two traps to avoid are under- and over-specification of variables to be monitored. In the former, you run the risk of neglecting or losing valuable information; in the latter your costs in data collection and analysis may overwhelm your best intentions.

Depending on the objectives of your prescribed fire, you could construct an almost endless list of variables to be measured. For example, you may be concerned with fire effects on vegetation, wildlife habitat, range condition, or soil and water resources. At the same time, depending on your past experiences, you will need to track influences of fuel, weather, topography, season, and ignition pattern on the spread and intensity of your fire treatment. Variations in spread, intensity, and post-flaming attributes will produce differential effects on plant injury, mortality, and survival; fuel consumption; and duff removal and bare mineral soil exposure. Your challenge is to come up with a stingy list of variables which will enable you to evaluate the quality of your efforts, replicate or learn from for future applications, and which you can afford to measure, given the wide array of techniques and procedures which have been developed by researchers and managers.

This task is all the more formidable since few standard techniques for conducting monitoring and evaluation have been published. This gap is largely due to the fact that no guidelines will apply in all situations. Fischer (1978), Martin and Dell (1978), and Ryan and Noste (1985) provide useful guidelines for evaluating prescribed fires. Other useful approaches for documenting fire behavior include Rothermel and Deeming (1983), and Rothermel and Rinehart (1983). Beyond these guidelines, you may find that your best allies are your own creative instincts and other resource specialists who are either cooperating with you in the conduction of a burn, or have confronted similar problems in other areas.

At this point, you may find yourself asking the question: Is it too costly to monitor and evaluate fire management decisions? My answer is that given what we don't know about prescribed fire effects, we can hardly afford not to carry out such activities.

Prescribed Fire Monitoring and Evaluation Practices

Several years ago, I collaborated on a study of prescribed fire reporting practices. The study was sponsored by the Rocky Mountain Forest and Range Experiment Station in order to examine current practices and to suggest future refinements. I'd like to briefly highlight some of our findings at this time, including those findings reported by a graduate student and me (Brandel and Omi, 1983).

To conduct our study, we sent a survey and request for prescribed fire reports to public agencies involved in prescribed burning activities throughout the West. Our findings are based on returns of 412 prescribed fire reports. Throughout the West, hazard reduction was most often cited as the objective of burning, with site preparation and wildlife habitat improvement next in importance. An exception to this ranking was reported in the Rocky Mountain region, where wildlife habitat improvement was the most often cited reason for burning.

Table 1 indicates the percentage of respondents who measured particular prescription variables, before, during, after a fire treatment. Note that vegetation descriptors were most often measured, but that fuel loading information often was not collected prior to burning. Further, loading information was even less

frequently reported after burning, especially by size class. These data suggest an important question: If fuel hazard reduction is so often cited as the reason for burning, how can practitioners truthfully assess the effectiveness of burning without taking pre- and post-burn measurements?

Table 1 also shows that weather variables are most often measured during a burn, although desired prescription weather may not be included in the report. A similar trend is reflected in fuel moisture measurements, although these are collected less often than weather measurements. Fire behavior descriptions are quite limited, with the exception of ignition pattern and type of burn. You have to conclude that either there is no need to write down desired vs actual conditions, or else future refinement is not necessary, either as a learning tool or device for communicating with others. While this may be the case in some applications, I doubt whether we can permit such laxity in reports until all uncertainty is eliminated about using fire in a particular vegetation type. Further, in the absence of accessible documentation, much of the valuable experience gained in conducting burns will disappear when the current expert retires or transfers job responsibilities.

A question logically raised by analysis of table 1 relates to how practitioners can assess the success of burns, especially if measurements are not being collected to enable pre-burn and post-burn comparisons. Success is usually defined in terms of objective accomplishment. Table 2 suggests other possible interpretations may be in use. One criterion apparently invoked on some burns relates to the ease with which an area can be ignited. As suggested in table 3, burners are less able to achieve objectives when sustained ignitions cannot be easily maintained. Thus a fire which is difficult to ignite can be considered a failure, just as one which escapes control. Similarly, fires which are difficult to control may be considered successful, irrespective of their long-term effects on the resources. Although such a "scorched-earth" burn policy is hardly a sufficient indicator of treatment success, practitioners may not have the luxury of relying on other indicators, especially if fulfillment of targets override more rational considerations.

Suggested Refinements

Based on our case study, we proposed items to be considered in reporting prescribed burns (tables 3-6). Table 3 outlines major steps to be included in a prescribed burn report. These include location/burn unit description, burn objectives, monitoring considerations, a narrative, and evaluation. Key points include the need to quantify burn objectives; and to identify post-burn evaluation date, fire descriptors (table 4), and site and environmental measurements to be collected (tables 5 and 6). The key feature of tables 4 and 6 is the need to refine prescriptions based on planned versus actual fire behavior and environmental conditions; table 5 requires comparisons between pre-burn, desired, and post-burn site and resource measurements. Filling in the last column in tables 4-6 forces the manager to think about future refinement, or to go out after the burn (perhaps with a resource specialist) to actually measure outcomes from the burn. These assessments are vital to improving the quality of future efforts. Mention importance of future refinement to improve quality in tables 4-6.

Adoption of standard reporting procedures would facilitate information exchange. Ideally, a common reporting format could be developed, preferably region-specific within an agency. For example, the USFS, BLM, NPS, and State have developed a common, shared data base in Colorado.

Table 1—Variables included in fire reports (n = 412) and percentage of respondents reporting measurements on each variable (Brandel and Omi 1983).

Variables	Percentage
Vegetation	
Overstory Vegetation	74
Understory Vegetation	73
Fuel Loadings	
Pre-burn loading by size classes	[1]8
Pre-burn total fuel loading	49
Post-burn loading by size classes	[2]9
Post-burn total fuel loading	21
Fuel Moistures	
Desired fuel moisture by size class	[3]24
Actual fuel moisture by size class	[4]27
Weather	
Desired weather conditions	[5]63
Actual weather conditions	[6]84
Fire Behavior Indicators	
Ignition pattern	67
Type of burn	42
Actual fire behavior	[7]26
Actual NFDRS indexes	[8]17

[1]*Average percentage of respondents reporting preburn loadings for 1-hr (18), 10-hr (19), 100-hr (19), 1000-hr (15) fuels, and duff depth (17).*

[2]*Average percentage of respondents reporting post-burn loadings for 1-hr (9), 10-hr (9), 100-hr (9), 1000-hr (9) fuels, and duff depth (7).*

[3]*Average percentage of respondents reporting moisture contents for 1-hr (27), 10-hr (50), 100-hr (12), and 1000-hr (8) fuels.*

[4]*Average percentage of respondents reporting actual moisture contents of 1-hr (32), 10-hr (52), 100-hr (15), and 1000-hr (10) fuels.*

[5]*Average percentage of respondents reporting desired temperature range (61), relative humidity range (71), wind direction (48), and wind speed range (72).*

[6]*Average percentage of respondents reporting actual temperature range (86), relative humidity range (92), wind direction (68), and wind speed range (90).*

[7]*Average percentage of respondents reporting observed maximum flame length (21), average flame length (28), and rate of spread (30).*

[8]*Average percentage of respondents reporting actual Ignition Component (19), Spread Component (11), Energy Release Component (16), and Burning Index (21) of the National Fire Danger Rating System.*

Table 2.—Number of times objectives are accomplished,[1] as affected by burnability.

(x^2 = 21.06 with 6 degrees of freedom, $p < .05$)

Objectives	Difficult to control	Burns without help[2]	Difficult to burn[3]	Won't burn
Completely met	9	83	5	0
Partially attained	0	13	3	0
Not met	0	0	2	3

[1]From data summaries for hazard reduction, site preparation, and wildlife habitat improvement objectives only.

[2]Additional ignition efforts unnecessary to sustain combustion.

[3]Sustained combustion impossible without additional ignition efforts.

Table 3.—Suggested prescribed burn report format.

I. LOCATION/BURN UNIT INFORMATION
 LEGAL DESCRIPTION, SIZE, VEGETATION TYPE, PROJECTED COSTS

II. BURN OBJECTIVES
 QUANTITATIVELY STATED

III. MONITORING
 POST-BURN EVALUATION DATE
 FIRE DESCRIPTORS (table 4)
 SITE/RESOURCE MEASUREMENTS (table 5)
 ENVIRONMENTAL CONDITIONS (table 6)

IV. NARRATIVE

V. EVALUATION
 ASSESSMENT OF ACCOMPLISHMENTS (OBJECTIVES)
 SPECIAL PROBLEMS
 COSTS
 BENEFITS
 FUTURE REFINEMENTS

Table 4.—Fire descriptors to be monitored and evaluated, and future prescription refinements.

	PLANNED	ACTUAL	FUTURE REFINEMENT
FUEL MODEL			
FIRING PATTERN			
STRIP WIDTH			
ROS			
FLAME LENGTH			
RESIDENCE TIME			
SCORCH HEIGHT			

Additional Comments

Monitoring Techniques

Some final comments are appropriate with respect to selection of appropriate monitoring techniques. Several different techniques may apply, depending on your objectives and circumstances. Typically, the following comparisons apply in fire ecology studies.

1. **Burn vs. unburned:** most common, site similarity assumed, fire behavior may or may not be known
2. **Postburn vs. preburn:** do not need to assume similarity, but fire behavior still may not be documented
3. **Preburn, on-site fire behavior, and postburn measurements:** essential for research burns, replication of experiments, and quantification—however may not be feasible in operational burns.

As you might guess, the strength of your inferences will vary considerably, depending on the type of above comparison you are making. In the first two instances, you must assume that the severity of burn can be used to estimate the type of fire which occurred in the area. Too often, inexperienced fire researchers do not realize that different fire intensities and duration of burning will dramatically alter post-fire effects. Consultation with other specialists may be needed to select the most appropriate sampling design, number of samples, and data collection and analysis methods.

Evaluation Techniques

There are no standard techniques for conducting evaluations, although guidelines may be in existence for your agency. Important items to be

addressed in any evaluation include assessment of accomplishments, fire effects on the environment and resources, effectiveness of the prescription, costs, and future refinements.

Several agency report formats include space for narrative assessment of objective accomplishment, special problems, and future refinements. The comparisons embodied in tables 5-6 facilitate comparison of the burn with intended or planned outcomes. Where significant discrepancies exist, an objective evaluation is valuable in terms of identifying possible causes and future remedies. Other items which might be pointed out in an evaluation include accidents, near accidents, public responses, or additional comments received from interested observers.

A thorough evaluation will generally consider issues related to two broad subject areas: management of the fire, including logistics; and changes in the resource(s) induced by the fire treatment. Ideally, the prescribed fire specialist will interact with both the burn boss and resource specialist in reviewing and evaluating fire outcomes from these two perspectives. Commitments for re-evaluations in the future may also be necessary.

Summary

Although I have discussed many factors to be considered if you are developing monitoring and evaluation plans, several are probably most important. These include: Resource and fire management objectives; costs; your previous experiences; and existing plans, information networks (including fellow burners), available data bases and published literature.

Before embarking on a costly and time-consuming monitoring and evaluation plan, you need to ask yourself a few simple questions with respect to prior burns. First, are you satisfied with the progress to date in terms of achievement of overall goals and objectives? If so, are improvements possible? If not satisfied, where are the areas most in need of improvement? Your straightforward answers to these questions will help you determine the appropriate course of action.

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Table 5.—Site/resource measurements to be monitored and evaluated.

	PREBURN	DESIRED	POST-BURN
FUELS/RESIDUES (size class, dead/live)			
LIVE TREES (spp/sizes)			
SHRUBS (cover, density frequency)			
HERBACEOUS VEGETATION (spp, cover, load)			
SNAGS (size classes)			
MINERAL SOIL Exposure (%)			
PLANTABLE SPOTS (#/ac)			
INSECTS/DISEASE (% trees affected)			

Table 6.—Environmental conditions to be monitored and evaluated.

	PLANNED	ACTUAL	FUTURE REFINEMENT
TEMPERATURE			
RELATIVE HUMIDITY			
MIDFLAME WIND			
WIND DIRECTION			
SHADING			
1 HR FUEL MOISTURE			
10 HR FUEL MOISTURE			
100 HR FUEL MOISTURE			
1000 HR FUEL MOISTURE			
LIVE FUEL MOISTURE			
DUFF MOISTURE			
SOIL MOISTURE			

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Jues

Opportunities for Fire Management in the Future[1]

Peter F. Ffolliott[2]

Abstract.—Fire, either as a natural occurrence or a management tool, can have beneficial effects on the environment, and its use offers opportunities for reducing fuel loads, disposing of slash, preparing seedbeds, thinning stands, increasing herbaceous plant production, increasing streamflow, and creating esthetic environments. Fire has been used for these purposes, to various extents, in southwestern ecosystems for decades. A review of studies on the effects of fire is presented, along with a discussion on related environmental, economical, and educational factors.

Fire has been an integral part of the ecology of southwestern ecosystems. The effect of fire in these ecosystems has been beneficial or detrimental, depending upon the nature of the fire, the characteristics of the fire site, and the values of the natural resources affected by the fire. In this paper, the opportunities for the use of fire as a tool to satisfy a management objective in southwestern ecosystems will be explored through a review of the literature.

The meeting of a management objective often can be achieved by manual or mechanical methods, although the costs of applying these methods may be limiting. A management objective also might be attained through the use of chemicals, but in this case, application may be restricted by legislation or regulation. An alternative means of satisfying a management objective in southwestern ecosystems may be the application of fire, when this application is environmentally sound, economically justifiable, and publically supported.

In this review of opportunities for fire management, three groupings of vegetative communities have been considered, specifically, coniferous forests, pinyon-juniper woodlands and chaparral, and desert shrub and grassland communities. More particularly, the opportunities for fire management in these vegetative communities have been considered in terms of a reduction of fuel loads, disposal of slash, preparation of seedbeds, thinning of stands, increased production of herbaceous plants, improvement of habitats for wildlife species, response of streamflow, and the esthetics of fire. All of these opportunities do not necessarily occur in all of the vegetative communities considered, however.

The focus of this paper has been placed on the opportunities for prescribed fire management. However, in reviewing the literature, it has become apparent that the effects of controlled burn treatments, and vegetative-modifying and vegetative-replacing wildfires also can be important in imputing the potential effects of prescribed fire in southwestern ecosystems. Therefore, information from these sources has been included in the paper.

CONIFEROUS FORESTS

Fire has been used successfully throughout much of the range of coniferous forests in the southwestern United States to reduce fuel loads, dispose of slash after timber harvesting, prepare seedbeds for natural regeneration, and in some instances to thin forest stands, and in other instances to stimulate the production of herbaceous plants (Lotan et al. 1981). Opportunities also appear good for utilizing fire to improve habitats for wildlife species and, in theory, to increase streamflow. Fire always has been part of the ecology of these coniferous forests (Weaver 1951, Arnold 1963, Zwolinski and Ehrenreich 1967, Biswell et al. 1973, Dieterich 1980, Dieterich 1983), and probably always will be.

Reduction of Fuel Loads

A review of the literature reveals a limited amount of information on the effects of fire on fuel loads in southwestern coniferous forests. Most of the reported work has been in ponderosa pine forests, because of the economic importance, the frequent wildfire occurrence, and the fact that prescribed fire has been used relatively extensively as a management tool in the type (Martin et al. 1979). Little information is available for the mixed conifer forests (representing a variety of intermixtures including Engelmann spruce, blue spruce, Douglas-fir, white fir, corkbark fir, ponderosa pine, and southwestern white pine, with a scattering of quaking aspen), although this type does represent a forest ecosystem in which studies on fire effects are needed.

Earlier papers by Weaver (1951), Biswell et al. (1973), and others de-

[1]*Panel paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Professor, School of Renewable Natural Resources, College of Agriculture, University of Arizona, Tucson, Arizona 85721.*

scribed the importance of fire in the ecological change of southwestern ponderosa pine forests. Of particular note in these descriptions was the fire prevention and suppression activities of land management agencies from the period of the early 1910s and 1920s through the late 1960s and early 1970s, that allowed large amounts of fuels, including logs, limbs, bark, leaves, needles, and slash, to accumulate in the forests. Although the use of controlled and prescribed fire in more recent years has helped to reduce the volumes of dead, highly flammable fuels, there undoubtedly are additional opportunities to further this work.

The role of fire in reducing the flammability of ponderosa pine forests through the consumption of excessive surface fuel volumes has been recognized for some time (Weaver 1951, Kallander 1958, Knorr 1963, Kallander 1969, Biswell et al. 1973, Wagle and Eakle 1979). The work by Knorr (1963), who analyzed individual fire reports on the Fort Apache Reservation, in eastern Arizona, from 1953 through 1961, illustrates this point.

From his study, in which 1,852 wildfires in areas not burned by controlled fire and 439 fires in controlled-burn areas were compared in terms of the size of the subsequent wildfires on the two areas, Knorr (1963) estimated that the reduction in the average size of the fires in the treated area was approximately 60%. It also was concluded that the number of wildfires had been reduced by the controlled burning treatments.

To further illustrate the use of fire on fuel buildups in southwestern ponderosa pine forests, additional examples of the effectiveness of fire in consuming fuels, considering both dead and down woody materials and the dead organic materials on the forest floor, are presented in the following paragraphs. Parenthetically, the combined litter (L), fermentation (F), and humus (H) layers of dead organic materials that accumulate on the forest floor will be collectively referred to as litter, for simplicity.

A study of the effects of a controlled fire on fuels in ponderosa pine forests, also conducted on the Fort Apache Reservation, revealed a reduction in the litter beneath the burned forests stands from about 30.5 to 26 tons per acre (Cooper 1961). Only in openings, where the litter is shallow in depth and discontinuous, was bare soil exposed following the burn.

In another study in the same general area, however, it was reported that "major changes" in fuel loads after another controlled fire were found only on about 25% of the area that burned by hot surface or crown fires (Lindenmuth 1960, Lindenmuth 1962). It was concluded in this latter study that more knowledge of desired fire intensities to meet fuel reduction objectives and practical techniques for controlling fire intensities still was limited.

In an effort to determine the amount of fuel consumed by a wildfire on the Apache National Forest (Dieterich 1976), paired plots were established in a ponderosa pine stand that was bisected by the fire. Dead and down woody materials amounted to 42 tons per acre and 10 tons per acre of litter on the unburned plot (Martin et al. 1979). On the adjacent burned plot, the measurements indicated that 13 tons of the large woody fuel and 9 tons of litter were consumed by the fire. Of these 22 tons, it was estimated that from 8 to 12 tons of fuel were consumed in the fire front.

A prescribed fire was set in a ponderosa pine forest near Flagstaff, Arizona, to burn approximately three-fourths of the litter depth. This objective generally was satisfied (Davis et al. 1968), although 11 years after the burn, compaction of the initial needle drop and the normal needle fall created a litter depth that was two-thirds the pre-fire level (Ffolliott et al. 1977). It was concluded that periodically prescribed fire would be necessary to maintain the reduction in litter amounts.

A controlled fire on the Fort Apache Reservation one year before a catastrophic wildfire occurred effectively reduced the impacts of the subsequent wildfire on the ponderosa pine forest overstory, the surface vegetation, and the organic layers in the soil profile (Wagle and Eakle 1979). Presumably, the consumption of fuels by the control fire was the cause, suggesting that controlled or prescribed fire can be an effective tool in alleviating the serious effects of wildfire in southwestern ponderosa pine forests.

In more recent years, fire prescriptions have been tested in an attempt to meet a specific objective of reducing loads in ponderosa pine forests (Davis et al. 1968, Sackett 1980, Harrington 1981, Harrington 1987a). One result of these tests has been the formulation of predictive equations that indicate that the consumption of naturally occurring fuels can be estimated from readily obtained variables. The work by Harrington (1987a) exemplifies this work. In essence, his study indicated that fuel consumption in ponderosa pine forests in southeastern Arizona can be estimated from knowledge of the moisture content of the H layer, and either pre-fire litter depth or forest stand density. These equations present foresters with an ability to predict and, therefore, prescribe the amounts of fuel that would be consumed by prescribed fire or unplanned ignition.

Disposal of Slash

The addition of fuels created by timber harvesting or pre-commercial thinning to the naturally occurring fuels often is a major concern to foresters. These additional activity fuel loads increase the fire hazard drastically. In a study of alternative fuel treatments in southwestern ponderosa pine forests, considering only

timber-related cash flows, it was determined that piling and then burning the slash is the most effective and, therefore, preferred fuel treatment for these forests (Hirsch et al. 1979). However, a complete analysis of fuel treatments must involve economic consideration of the effects of fuel treatment on fire hazard, timber production, wildlife habitat, soil characteristics, water yield and quality, and recreational values.

The analytical procedures outlined in the study by Hirsch et al. (1979) provide foresters with the best available information concerning the fire hazard aspects of activity fuel treatment (at the time of the analysis). In addition, the procedures document the factors involved in reaching a decision on the selection of a treatment.

Preparation of Seedbeds

Fire has potential value in preparing seedbeds in southwestern ponderosa pine forests for natural regeneration. Conditions affected through burning that favor the germination of seeds and subsequent survival and early growth of seedlings include a more "receptive" seedbed because of the removal of litter and exposure of mineral soil, increased nutrient availability, and more favorable conditions of soil moisture and temperature.

More seedlings started on areas subjected to a prescribed fire in a ponderosa pine forest near Flagstaff, Arizona, than on adjacent unburned areas (Davis et al. 1968), although many of these seedlings were short-lived (Ffolliott et al. 1977). One year after the fire, the newly established seedlings occupied 90% of the mil-acre plots on the burned areas, compared with 16% on the adjacent unburned plots. However, established seedlings occupied only 25% of the plots on the burned areas 11 years after the fire. No plots adjacent to the burned areas still supported seedlings at that time.

Observations of natural regeneration after the initial prescribed burns at Chimney Spring, near Flagstaff, Ariz. (Dieterich 1980), further supported the thought that prescribed fire can prepare ponderosa pine seedbeds for regeneration, and that if the prescribed burn is repeated, the seedbeds will remain receptive to regeneration for several years (Sackett 1984). Removal of the heavy litter accumulations, exposing mineral soil, releasing nutrients bound up in the litter, improving moisture conditions, and warming the soil surface were believed to be the causes. Although a large percentage of the original seedlings died as a result of drought, frost heaving, and animal damage, a conclusion of this study was that prescribed burns in southwestern ponderosa pine forests can be a cost-effective method of preparing seedbeds (for natural regeneration) in openings that are too small for the seedling plantings.

In another study, prescribed burn also prepared a more favorable seedbed for the germination of seed and early seedling establishment in a ponderosa pine forest on the Fort Valley Experimental Forest, northwest of Flagstaff, Arizona (Haase 1986). The more favorable seedbed was achieved primarily by exposing mineral soil and increasing soil moisture, resulting in a nearly a twenty-fold increase in the number of seeds germinating on burned as compared to unburned sites. Birds, rodents, and drought subsequently took a heavy toll, however; only 8% of the seedlings, all of which were protected with wire cones, remained alive at the end of the one year study.

A good seed crop, in addition to the preparation of a seedbed, is required for successful natural regneration in southwestern ponderosa pine forests (Sackett 1984). Therefore, in prescribing fire to prepare a seedbed, potential seed trees should not be severely damaged. Fortunately, fire-scorched ponderosa pine trees, as distinguished from trees where the foliage has been consumed, have a reasonably good survival rate, depending upon the time and intensity of the fire and the extent of scorching. Furthermore, as reported in a study conducted near Flagstaff, Arizona, while cone size, seed soundness, and seed weight were less in trees whose live crowns were scorched as much as two-thirds, these trees still produced highly viable seeds (Rietveld 1976). Seedfall from foliage-scorched trees apparently represents a significant seed source for the regeneration of burned areas.

One possible problem in the use of fire to prepare seedbeds is found on sites where Gambel oak occurs in intermixtures with ponderosa pine. Intensive surface fires effectively kills the tops of oaks, but the oaks subsequently sprout, often profusely, after the top kill. On areas in eastern Arizona, intense surface fires have released the oak to the point where ponderosa pine trees failed to become re-established (Biswell et al. 1973).

Ponderosa pine seedlings have been planted in some areas with reasonable success after a fire removed the competing vegetation and prepared the sites for planting. To illustrate this point with a case study, the survival of planted seedlings one growing season after a wildfire near the West Fork of Oak Creek, in north-central Arizona, was 80% (Campbell et al. 1977). After the second growing season, the survival rate still was 68%. Importantly, stock from seed of local origin was recommended for these post-fire plantings.

Thinning of Stands

Southwestern ponderosa pine stands often become stagnated and overstocked with the smaller size classes. Overstocking in these smaller size classes can be overcome by manual or mechanical means, although the costs may be high and the ques-

tion of slash disposal must be answered. Therefore, the use of fire to thin stands, especially from below, has been considered and, in many such instances, prescribed. The effectiveness of fire in the thinning of ponderosa pine stands can be illustrated through a review of a few studies.

In his study of the effects of a low-intensity controlled fire in a ponderosa pine forest on the Fort Apache Reservation, Lindenmuth (1960, 1962) reported that about 25% of the potential "crop trees" needing release from the understory stands, in fact, were released by the fire. However, for every crop tree released, an average of nearly 6 potential crop trees required to utilize the existing growing space were either damaged or killed. This latter result reinforced a conclusion of this study that more knowledge was necessary to prescribe fire to meet specific thinning objectives.

Analysis of a controlled fire in a ponderosa pine forest near Flagstaff, Arizona, also showed that potential crop trees could be released by burning (Davis 1965). Because the fire intensity was low, only a small proportion of the crop trees needing release were released, however. This contolled burn and that described by Lindenmuth (1960, 1962) were similar in terms of fire intensities and in the proportions of the areas burned by light-surface, hot-surface, and crown fires.

The results of still another study of a controlled fire on the Fort Apache Reservation indicated that the effectiveness of fire in the thinning of ponderosa pine trees was related, in large part, to the concentrations of fuels on the ground before the fire (Kallander 1969). On some sites, a "satisfactory" thinning was achieved, while the density of trees on other sites was reduced only slightly. The "uneveness" of the treatment was attributed, in most instances, to the spatial distributions of heavy fuels on the ground.

A severe wildfire near the Fort Valley Experimental Forest decimated an unthinned ponderosa pine stand, while an adjacent thinned stand was unchanged relatively (Pearson et al. 1972). A crown fire eliminated all trees in the unthinned area, which originally supported 135 square feet of basal area per acre; the ground fire in the thinned area, which had been reduced to about 25 square feet of basal area per acre before the fire, caused no significant loss of trees. The results of this study suggest a "threshold" level of forest density at which or below which a ground fire will have little effect in the thinning of ponderosa pine trees.

An evaluation of changes which resulted from the wildfire near the West Fork of Oak Creek provides further insight into the effects of fire on a southwestern ponderosa pine forest (Campbell et al. 1977). Here, nearly two-thirds of the forest overstory was destroyed on a severely burned area (with an estimated fire intensity of 10,000 BTUs per second per foot), with mortality the greatest in the smaller size classes. On a moderately burned area (with an estimated fire intensity of 2,500 BTUs per second per foot), less than 30% of the trees were lost, again, mostly in the smaller size classes.

Attempts to prescribe fire to meet, at least partially, the objectives of thinning stands have been undertaken in southwestern ponderosa pine forests. For example, the prescribed fire set (near Flagstaff, Arizona) to consume three-fourths of the litter depth also thinned the stands from below (Davis et al. 1968). In excess of 45% of the basal area was lost to burning on one site, a reduction of 170 to 90 square feet per acre. On another site, nearly 25% of the basal area was lost, from 305 to 235 square feet per acre. The majority of the trees killed were less than 6 inches dbh on both areas. Eleven years after the fire, the basal area on the former site had increased to 120 square feet per acre, indicating a growth rate of 3% annually (Ffolliott et al. 1977). The basal area on the latter site actually decreased, to 210 square feet per acre, 11 years after the prescribed burn, suggesting that enough trees initially damaged by the fire subsequently died to offset the growth of residual trees.

Many stagnated and usually smaller ponderosa pine trees were thinned from below from sites initially burned at Chimney Spring (Sackett 1980). Although this prescribed burn was considered to be a "good start" in reducing the density of the overstocked stands, it was concluded that it would take many additional prescribed fires to make these stands productive.

In his study of prescribed burning in ponderosa pine forests in southeastern Arizona, Harrington (1981) reported that the smallest trees were killed in most instances, increasing the average tree diameters in the residual stands. Once again, the greatest tree mortalities were associated with areas of greatest fuel reductions. On sites characterized by large, old growth ponderosa pine trees, the basal areas decreased only slightly, because these larger trees were not affected by the fires. However, the basal areas were reduced significantly on sites supporting dense, overstocked clumps of saplings, commonly called "dog-hair-thickets," because of the high mortality rates.

When consideration is given to the use of fire to thin stands, attention also must be directed toward the recovery of fire-damaged trees prescribed to remain in the stands. Only rarely will no trees be damaged by fire. Therefore, the recovery potential of fire-damaged trees should be incorporated into the planning of prescribed fire to thin stands. The survival potential of fire-damaged ponderosa pine trees generally is dependent upon many factors, including the season when the fire occurs, the percentage of crown scorch, and the consumption of live crown (Di-

eterich 1979, Harrington 1987b). Other factors influencing the survival of these trees are site conditions, available growing season moisture, and subsequent insect attacks.

Increased Production of Herbaceous Plants

In a number of studies, investigators have reported immediate stimulation of herbaceous plant growth through burning, resulting in greater forage production and forage yields (Lyon 1974). Soils generally are warmer on burned areas and, therefore, spring plant growth starts earlier. Soil fertility usually is increased by the release of nutrients, also encouraging the growth of herbage (Biswell et al. 1973). Plant vigor often is promoted by the removal of senescent shoots and foliage and, in many instances, burning of the litter prevents the interception of light and water, once again, favoring plant growth. Opportunities for stimulating the production of herbaceous plants in southwestern ponderosa pine forests are illustrated through a review of previous studies on this potential effect of fire.

Herbage production increased from less than 5 to 40 pounds per acre 1 year after the prescribed fire was set to consume three-fourths of the litter in a ponderosa pine forest near Flagstaff, Arizona (Davis et al. 1968). However, most of this increase was attributed to the production of relatively unpalatable plants, with grasses, grass-like plants, and browse remaining unchanged. Herbage production also was approximately 40 pounds per acre 11 years after the fire, but the herbage composition had changed greatly (Ffolliott et al. 1977). The unpalatable plants had been replaced by a mixture of perennial grasses and forbs preferred by livestock and wildlife species.

The severe wildfire near the Fort Valley Experimental Forest initially stimulated the growth of herbaceous plants in both the unthinned and thinned ponderosa pine stands (Pearson et al. 1972). Herbage production continued to increase in the second growing season after the fire in the unthinned area, where the greatest reduction in the forest overstory occurred, while it decreased in the thinned area. The nutrient values of the herbaceous plants, including crude protein, phosphorus, and in vitro digestibility, were enhanced temporarily after the burning.

There were no significant differences in herbage production on severely burned, moderately burned, and unburned areas in and adjacent to the wildfire in the ponderosa pine forest near the West Fork of Oak Creek at the end of the initial growing season after the fire (Campbell et al. 1977). However, at the end of the second growing season, herbage production on the severely burned and moderately burned areas had increased threefold, from 500 to nearly 1,500 pounds per acre, in comparison to the unburned area. At this time, the herbage composition on the moderately burned area had approached that of the unburned area, while the herbage composition on the severely burned area continued to show successional changes.

Herbage production also was measured 9 years after the wildfire near the West Fork of Oak Creek, with striking changes reported (Oswald and Covington 1983). There was a significant decrease in herbage production on the severely burned area, in comparison to the measurements taken 2 years after the fire, with a greater relative decline in the production of preferred forage species. This phenomenon was caused, in large part, by increased occurrence of unpalatable plants, related apparently to the heavy grazing by livestock on the severely burned site. Herbage production and the production of forage plants did not decrease significantly in the period between 2 and 9 years after the wildfire. Because of the wide variation in the responses of herbage plants, Oswald and Covington (1983) advised against making generalizations on the effects of fire on herbage production in southwestern ponderosa pine forests.

In a study of changes in herbage production for three different prescribed burns of different ages (2, 5, and 7 years) in southwestern ponderosa pine forests, it was reported that herbage production for 1 to 2 years following the burns exhibited either no change or an increase (Andariese and Covington 1986). The production of preferred forage plants for the same period showed either no change or a decrease. However, responses in herbage production and the production of forage plants more than 2 years following fire showed a general trend of increasing levels. It was concluded, therefore, that prescribe fire in southwestern ponderosa pine forests (and on basalt soils, as studied here) can produce long-term benefits in increased production of herbaceous plants and forage species, especially in the pole-size stands which dominate the region.

In addition to measurable increases in herbage production, greater forage availability has been reported, where unpalatable plants become palatable after a fire, where physical barriers to plant utilization by livestock and wildlife species have been burned, and where plants too large for utilization have been reduced in size by burning (Lyon 1974).

Improvement of Habitats for Wildlife Species

One of the major problems in attempting to document the effects of fire on wildlife species is that few studies on this topic are quantitative, have adequate controls, or have been carried on long enough to properly analyze the effects on the wildlife species (Lyon et al. 1978). Another complicating factor is that little infor-

mation is available to describe the effects of fire on the dynamics of wildlife populations. Much of the information about fire effects on wildlife species describes plant community modifications by fire, and the consequent influences on the food and cover utilized by wildlife species.

In this review paper, the direct effects of fire on the dynamics of wildlife populations has been separated from the habitat modification effects on these populations, with the following discussion focused on the latter. Food, cover, and the environment in general can be modified by the disturbance of fire, and for a considerable number of years thereafter, plant succession will continue to produce substantial changes (Lyon et al. 1978). That these modifications and changes can impact on wildlife species in southwestern ponderosa pine forests, often beneficially, has been recognized generally (Jones 1973, Wagle 1981), and can be illustrated specifically with a review of several study results. Importantly, in these studies, observations and measurements of the presence of wildlife species in a particular habitat have been considered to reflect that species relative preference, or use, for that habitat.

Following the wildfire in a ponderosa pine forest near the Fort Valley Experimental Forest, elk and deer pellet groups were counted to indicate the relative use by these big game species of burned and unburned areas (Kruse 1972). It was found that forest openings created by the fire and followed by seeding with perennial grasses and forbs were just as attractive to elk as were open habitat conditions created by the clearcutting of the forest overstory and subsequent seeding. In fact, elk use of the burned areas was higher than on either the clearcut or adjacent thinned areas for 2 years following the fire. However, the third year of the study showed an "equalizing trend" in elk use among all habitat conditions studied. Deer use also increased on the burned areas, with a steady increase in the number of pellet groups counted annually in the 3 years following the fire.

Changes in animal life after the wildfire near the West Fork of Oak Creek provides further evidence on the effects of fire on wildlife species in a ponderosa pine forest. Deer use on the moderately burned area, again, indicated by pellet group counts, was greater than on the unburned areas in all 3 years of measurements after the fire (Campbell et al. 1977). However, only in the third year was deer use on the severely burned area greater than on the unburned area. The pattern of deer use in the different years appeared to be related to the production of palatable herbaceous species (Neff 1974). Unfortunately, the counts of elk pellet groups were confounded by management changes for cattle in the area. Elk and cattle often compete for food and space on grazing lands in southwestern ponderosa pine forests; therefore, it is difficult to isolate the effects of a disturbance on one animal without considering the other.

Changes in rodent populations, which frequently "index" changes in large herbivorous populations, also were studied by Campbell et al. (1977). Of the seven rodent species caught by live trapping, mice, ground squirrels, and chipmunks were the most numerous. More mice and ground squirrels and fewer chipmunks were trapped on the burned areas than on the unburned areas in the first year after the fire. The increase in mice was attributed to increased food, including herbaceous plants, seeds, and insects; the increase in ground squirrels probably was in response to the opening up of the forest overstory by the fire and the subsequent increases in grasses and forbs. Since chipmunks are more dependent upon trees than the other rodent species caught, their decrease was expected after the forest overstory was burned. Changes observed in the first year after the fire were not detectable in the following years of the study, suggesting that the effects of the fire on rodent populations were short-lived.

In many instances, the effects of fire on the habitat of a wildlife species can be evaluated only in the long-run, that is, over a period of several years after a fire; in other words, the modifications and changes in habitat conditions from fire is "felt" only years later. Such is the case with a number of the wildlife species found in southwestern ponderosa pine forests (Lowe et al. 1978). For example, depending upon the fire intensity, season when the fire occurred, and the nature of the habitat burned, elk and deer summer-fall use (as opposed to winter-spring use) can decline in the first few years after burning, only to increase significantly in later years, with the "net effect" of the fire being an improvement in habitat conditions. Conversely, tree-foliage-searching birds can increase in numbers in the first years after a fire, and then decrease to below pre-fire levels for the next several years and, perhaps, remain there into the future. It is important, therefore, that the opportunities to use fire to improve (or otherwise modify) the habitats for wildlife species in southwestern ponderosa pine forests, as well as in all southwestern ecosystems, be considered only on a "case by case" basis.

Response of Streamflow

Hydrologic processes that can be affected by fire include interception, infiltration, soil moisture storage, snowpack accumulation and melt, overland water flow, surface and mass erosion, and chemical quality of water (Tiedemann et al. 1979). Streamflow responses to the integrated effects of these hydrologic processes, in theory at least, are increased peak and total discharges, greater stormflows, and increased

baseflows. Unfortunately, only limited information is available to describe the effects of fire on hydrologic processes and streamflow from southwestern ponderosa pine forests.

A controlled fire that reduced the fuels in a ponderosa pine forest on the Fort Apache Reservation did not change the water-holding capacity of the litter appreciably (Cooper 1961). Therefore, from the results of this study, it seemed doubtful that fire (of the intensities studied at this time) would affect streamflows significantly. Nevertheless, Dieterich (1983) later hypothesized that carefully prescribed fires which reduced the litter depth and the density of the forest overstory could make more soil water available on a site which, in turn, could increase overland water flows.

Intense fires can decrease the infiltration capacities of forest soils (Zwolinski and Ehrenreich 1967, DeBano 1981). However, increased overland water flows are unlikely as long as the litter on the forest floor was not consumed completely. This point was supported, to some extent, by Clary and Ffolliott (1969), who, working in a ponderosa pine forest near Flagstaff, Arizona, determined that the well-decomposed H layer had to be modified or removed if the ability of the litter to intercept and then hold precipitation was to be reduced significantly. Such a modification or removal apparently happened as a result of a prescribed fire in a ponderosa pine forest on the Fort Valley Experimental Forest, where greater surface soil water was measured after the burn (Haase 1986).

There also has been little research on the effects of fire on streamflow from watersheds in southwestern ponderosa pine forests. However, two case studies, one based on a wildfire and the other on a prescribed burn, are summarized below.

Small instrumented watersheds, established on severely burned, moderately burned, and unburned areas in and adjacent to the wildfire in a ponderosa pine forest near the West Fork of Oak Creek, provided a basis to further study the effects of fire on streamflow (Campbell et al. 1977). Annual streamflows from the severely burned and moderately burned watersheds, which averaged 0.8 to 1.1 inches, were greater than those measured (0.2 inch) from the unburned watersheds. The casual factors were thought to be the burning of the litter to the mineral soil and the inducement of a water-repellent layer in the sedimentary soils. Streamflows declined in each of the two succeeding years of the study, however, and approached the prefire levels.

Average runoff efficiencies for these watersheds, defined as the ratio of surface runoff to precipitation, increased generally with the severity of the burn (Campbell et al. 1977). Runoff efficiencies for the three-year study averaged 0.8% on the unburned watershed to 2.8% and 3.6% on the moderately and severely burned watersheds, respectively.

In a paper presented elsewhere in these proceedings, Gottfried and DeBano (1989) reported that a prescribed burn, which covered nearly 45% of an instrumented watershed in a previously undisturbed ponderosa pine forest in eastern Arizona, did not increase streamflow significantly in the 6 years following the fire. This result was anticipated, though, because the damage to the forest overstory was minimal and the forest floor remained intact even though the surface fuels generally were consumed on the burned area.

Water quality parameters that are associated with streamflow regimes after fires also are important to foresters. In this regard, both the physical quality (sediment concentrations) and chemical quality of the water should be considered.

In his analysis of a controlled fire in a ponderosa pine forest on the Fort Apache Reservation, Cooper (1961) found that exposure of mineral soil and soil movement both increased by burning. However, most of the eroded soil materials only moved a short distance down the slope and, essentially, were stabilized in less than a year after the burning.

Movement of eroded soil materials following a wildfire in a ponderosa pine forest in the Sierra Ancha Mountains, near Globe, Arizona, also was limited (Rich 1962). During the first year after the fire, approximately 1 acre-foot of soil materials, an average depth of 0.016 foot, was eroded from the burned area of 60 acres. Most of these soil materials were deposited immediately below the burn, in unburned forest vegetation. Smaller amounts were deposited in the stream channel and in the weir pond. Only 2% of the eroded soil materials actually were trapped in the weir pond, although this amount was greater than measured before the fire.

In contrast to the above findings, in the first year after the wildfire in a ponderosa pine forest near the West Fork of Oak Creek, the soils on the severely burned watershed eroded greatly and moved into the stream channel; this produced a significantly greater suspended sediment yield, 1,254 pounds per acre, than the observed 3 pounds per acre on the unburned watershed. Suspended sediment yield from the moderately burned watershed was 44 pounds per acre. In subsequent years, the amounts of suspended sediments from the two burned watersheds decreased drastically.

In reference to the chemical quality of water, Sims et al. (1981) reported that the mean concentrations of calcium, magnesium, and fluoride were increased significantly after a controlled burn of small runoff plots in a ponderosa pine forest in southeastern Arizona. However, it was not determined in this study whether an increase in readily available nutrients on a burned site subsequently would increase the nutrient loadings of nearby streams.

A limited analysis of the streamflow from the small water-

sheds in and adjacent to the wildfire near the West Fork of Oak Creek indicated that the chemical quality of the water was not affected greatly by the burn (Campbell et al. 1977). In the initial runoff event after the fire, increased concentrations of calcium, magnesium, and potassium were measured. However, these concentrations declined in each of the subsequent runoff events, approaching the pre-fire levels in three years.

Changes in the chemical quality of water were analyzed before and after the prescribed fire on a watershed in the ponderosa pine forests of eastern Arizona (Gottried and DeBano 1989). This prescribed fire significantly changed the concentrations of some nutrients, namely nitrate-nitrogen, ammonium-nitrogen, phosphates, calcium, magnesium, sodium, and potassium, but these changes were too small to adversely affect water quality.

Esthetic Effects of Fire

On certain sites, controlled burns can improve the esthetics of a ponderosa pine forest by keeping the forest open and "park-like," and favoring the development of large, individual trees that can be viewed readily by the public (Biswell et al. 1973). Monotonous, dense, debris-litter clumps of saplings generally are uninviting both visually and physically.

The esthetic effects of prescribed fire in a ponderosa pine forest were analyzed in a case study near Flagstaff, Arizona (Anderson et al. 1982). The results of this study suggested that the esthetic effects, measured in terms of scenic beauty estimates, may depend upon the time frame of the evaluation. Immediately after burning, scenic values were lower than those for a comparable unburned area. Within one year, however, scenic values on the burned area surpassed those on the unburned area; and in subsequent years, the scenic values on the burned area essentially were equal to those on the unburned area. It was concluded that the "adverse scenic impacts" of prescribed fire were short-lived and, therefore, should not be a significant deterrent to prescribed burning in similar conditions.

PINYON-JUNIPER WOODLANDS AND CHAPARRAL

Fire frequently has been used to increase the production of herbaceous plants in pinyon-juniper woodlands and chaparral in the southwestern United States, to the benefit of both livestock and wildlife species (Arnold 1963, Zwolinski and Ehrenreich 1967, Lotan et al. 1981). Increasing streamflow is another potential benefit of fire in these vegetative types. There is little information on the use of fire in reducing fuel loads, although the conversion of pinyon-juniper woodlands and chaparral to a dominance of herbaceous plants can result in significant increases in herbaceous fuels.

Reduction of Fuel Loads

While there have been only a few studies on the effects of fire on fuel loads in southwestern pinyon-juniper woodlands, work by Arnold et al. (1964) on the effects of fire on plant succession, species compositions, and overstory density changes may be interpreted loosely in terms of changes in fuel volumes. It was found, for example, that wildfires occurring on flat to gently rolling terrain generally consumed available ground fuels, killed most of the trees, but left the "dead skeletons" of the trees standing. Ground fuels, rarely heavy in the pinyon-juniper woodlands, were estimated to be 1 to 3 tons per acre. In rougher terrain, where the continuity of the fuels was too sparse to support fire spread, islands of unburned trees were left on hills and ridges after burning.

A wildfire in a pinyon-juniper woodland on the Hualapai Reservation in northwestern Arizona effectively removed all of the litter cover, estimated to be 50% to 60% on adjacent unburned areas (Arnold et al. 1964). In addition, the overstory vegetation was destroyed, suggesting that planned fires "hot enough" to significantly reduce fuel volumes also might kill the overstory.

Little information is available to describe the effects of fire on fuel loadings in southwestern chaparral. In one study, however, Pase and Lindenmuth (1971), reporting on the effects of a prescribed burning program in chaparral in the Sierra Ancha Mountains, indicated that nearly 35% of the pre-fire litter, which averaged 6.4 tons per acre, was removed by burning. Importantly, under all of the burning conditions tested, a substantial amount of litter was left on the soil surface as protection against erosion.

Increased Production of Herbaceous Plants

The production of grasses, grass-like plants, and forbs generally is decreased with increasing densities of pinyon-juniper and chaparral overstories. Removal or reduction of these overstories, therefore, can increase the production of herbaceous plants. Fire, either naturally occurring or prescribed (often with seeding treatments), has been shown to remove or reduce pinyon-juniper and chaparral overstories and, therefore, increase the production of herbaceous plants, as described below.

A study of a wildfire in a pinyon-juniper woodland in south-central New Mexico indicated that the production of forage plants was reduced significantly, in comparison to an unburned area, in the year of the fire, but recovered by the end of the second year (Dwyer and Pieper 1967). Among the beneficial effects of the fire were the reductions in small

pinyon and juniper trees, and cholla plants, which subsequently should have led to an increase in the production of herbaceous plants. Another benefit of the burn was a release of nutrients for subsequent plant growth.

The production of perennial grasses and forbs increased dramatically following burning and seeding treatments in the pinyon-juniper woodlands on the Hualapai Reservation (Aro 1971). Specifically, herbage production on burned areas seeded with crested wheatgrass, western wheatgrass, weeping lovegrass, and yellow sweetclover was 1,660 pounds per acre, compared to 60 pounds per acre on adjacent unburned areas. In another large-scale burning and seeding program in the pinyon-juniper woodlands, Aro (1971) reported that the production of forage plants increased 500 pounds per acre.

The main purpose of an extensive prescribed burning and seeding program in the pinyon-juniper woodlands on the Hualapai Reservation from 1953 to 1963 was to provide increased forage resources for livestock production (Despain 1987). That this objective was satisfied is demonstrated by the reported increases in forage production on the treated areas. Depending upon the year and the site, estimates of forage production on the burned and seeded areas ranged from 200 to 2,200 pounds per acre, and on the adjacent unburned areas, from 25 to 280 pounds per acre. The burned areas still were dominated by the seeded forage species in 1986, with little invasion from pinyon and juniper trees.

Weeping lovegrass, seeded immediately after a wildfire in chaparral in the Pinal Mountains near Globe, Arizona, developed into a "good stand" during the initial growing season after the fire, and continued to increase in basal cover for 7 years thereafter (Pond and Cable 1962). Sprouting chaparral shrubs regrew rapidly at the same time, reaching a point 7 years after the fire that appoximated the shrub densities in unburned areas.

An objective of a prescribed burning program for chaparral on the Tonto National Forest in central Arizona was to burn the dense overstories, converting the area into "savanna-type" grasslands, but retaining islands of chaparral for wildlife cover (Courtney and Baldwin 1964, Baldwin 1968). Some of the results obtained from this program provide additional insights into the effects of fire in chaparral on the production of herbaceous plants.

Winter burning of a stand of Lehmann lovegrass, which caused a 90% top-kill of chaparral sprouts, had little effect on the density or vigor of this forage species, unlike the results reported for warm-season burns (Pase 1971). Increases were noted in the production of bluestem and native forb species, however.

In another study in the same area, Pase and Knipe (1977) further analyzed the effects of winter burning on forage species frequently utilized for seeding in southwestern chaparral. It was concluded from this study that the winter burning of weeping, Boer, and Lehmann lovegrass does not affect adversely the production of these species, and actually increases the production of forbs. The benefit to the forbs was due to a reduction in competition for light, resulting from the elimination of litter and standing vegetation. Furthermore, chaparral shrubs are top-killed by hot, flashy fires. Winter burning can be used to maintain a dominance by herbaceous plants in chaparral areas converted into lovegrass stands without danger to the grasses.

Herbaceous cover, almost absent before three small watersheds were treated with prescribed fire in the early fall in each of 4 consecutive years, increased greatly in the first year after the burns (Pase and Lindenmuth 1971). Herbaceous plants also made up a significant, but diminishing percent of the total cover 5 years after the burnings. Chaparral shrub canopies, which were reduced from nearly 60% to 3.5% by the burns, recovered to about 45% 5 years after the burns. The results of this study showed a good, but temporary control of chaparral overstories with carefully prescribed fire.

Chaparral sprouts are difficult to control with fire (Pond and Cable 1960, Lillie et al. 1964). It is important, therefore, to consider burning in chaparral overstories at relatively frequent intervals until the desired overstory control is achieved. In addition, the use of chemical desiccants in seasons of the year other than late spring and early summer, when the chaparral reaches its maximum level of flammability under natural conditions is recommended (Lindenmuth and Glendening 1962).

Improvement of Habitats for Wildlife Species

On areas converted from pinyon-juniper woodlands and chaparral by fire to increase the production of herbaceous plants, near complete removal of the overstories can eliminate the habitats for some wildlife species, while providing an improvement for others. When the burned areas are kept small and interpersed with pinyon-juniper and chaparral overstories, both protective cover and food will be available. Furthermore, the edge effect created by the burned openings generally enhances the overall wildlife environment.

Wildfires and prescribed burns to control pinyon and juniper trees can improve the habitats for deer populations, as reported in a study by McCulloch (1969) on the Hualapai Reservation. Total accumulations and rates of accumulation of deer pellet groups were greater in the burned than in the unburned parts of a long, narrow strip, 1/8 mile wide by 11 miles long, that included both burned and unburned areas. Despite the large acreage burned, only 5% of

it was more than 1/2 mile from unburned areas. It subsequently was hypothesized that small burns are desirable, if an area is to be managed only for deer, because small burns within unburned pinyon-juniper woodlands create a greater variety of cover and food than that available on burned or unburned large areas.

One finding of the prescribed burning program in chaparral on the Tonto National Forest was an improvement in the habitats for several wildlife species (Baldwin 1968, Hibbert et al. 1974). Opening up the dense overstories by fire apparently provided additional space for deer movements, and greater abundance and quality of browse and forage species. Increases in the quail and other bird populations after the burning treatments also were noted.

Response of Streamflow

A vegetative type conversion from a pinyon-juniper woodland to herbaceous plants, theoretically, should reduce the water loss from consumptive use and, in doing so, increase streamflow (Zwolinski and Ehrenreich 1967). Unfortunately, streamflow response to fire in the pinyon-juniper woodlands has not been measured adequately in the Southwest to verify this point.

It does appear, however, that streamflows from chaparral can be increased by removing the overstory by fire to establish a cover of perennial grasses and forbs (Glendening et al. 1961, Pase and Ingebo 1965, Pase and Lindenmuth 1971, Hibbert et al. 1974). A case study from the Sierra Ancha Experimental Forest in east-central Arizona illustrates this possibility.

Three years following the establishment of four small instrumented watersheds (B, C, D, and F) in chaparral on the Sierra Ancha Experimental Forest, all of the watersheds were burned by a wildfire. With the exception of C, all the watersheds were seeded with perennial grasses and forbs shortly after the fire. One year later, B was seeded again and C was seeded for the first time. Subsequently, control of the chaparral sprouts with herbicides was attempted on C and B, and streamflow responses (with D as a control) were measured (Hibbert et al. 1974). Chaparral sprouts on C were controlled through aerial applications and hand treatments with herbicides; annual streamflow increased on C nearly 6 inches as a result. Hand applications of herbicides to the chaparral shrubs on the north-facing slopes of B increased the annual streamflow by 1.2 inches. Aerial applications of herbicides on the total area of F also resulted in an annual streamflow increase of 2.7 inches. From this case study, it was concluded by Hibbert et al. (1974) that, if water is the major objective of management, the conversion of chaparral to herbaceous plants by fire and herbicidal treatments can be "wasted" unless three-fourths of the shrubs are eliminated.

Movement of eroded soil materials immediately after the wildfire and subsequent treatments with herbicides was greater than in the years before and after (Hibbert et al. 1974). However, measurements on C and B indicate that the conversion of chaparral to perennial grasses and forbs does not affect erosion any more than allowing the shrubs to recover. Furthermore, in the long run, this conversion should reduce erosion by eliminating the heavy erosion cycle set off by the periodic wildfires in unmanaged chaparral.

Stream water from the treated watersheds showed "moderate to low" contamination by the herbicides (Hibbert et al. 1974).

DESERT SHRUB AND GRASSLAND COMMUNITIES

The use of fire in the desert shrub and grassland communities of the southwestern United States has been limited largely to the control of shrub species and the subsequent conversion to perennial grasses and forbs (Humphrey 1963, Zwolinski and Ehrenreich 1967, Lotan et al. 1981). Such conversion treatments also can improve the habitats for wildlife species, most notably, bird populations (Bock et al. 1976). Opportunities to utilize fire as a practical management tool can be restricted, however, because the fuel loads often are inadequate to support a burning program (McLaughlin and Bowers 1982).

Increased Production of Herbaceous Plants

The effects of fire on shrub species and herbaceous plants in desert shrub and grassland communities can be illustrated through a review of some studies on these effects, as presented below.

In a study of the effects of fire on shrub and perennial grass species in a desert shrub and grassland community on the Page-Trowbridge Experimental Ranch, northwest of Tucson, Arizona, it was found that a prescribed burn effectively reduced the densities of cholla, burroweed, snakeweed, and Lehmann lovegrass (Humphrey and Everson 1951). Although the fire killed some of the Lehmann lovegrass plants, the balance of the affected stand was growing vigorously 1 year after the burn. Furthermore, as Lehmann lovegrass was established easily on burned sites, it was concluded that a "temporary" reduction in the stand had little permanent consequences, when compared with the benefits obtained from the control of the shrubs.

The effects of prescribed burning in desert shrub and grassland communities on the Santa Rita Experimental Ranch in southern Arizona was reported by Reynolds and Bohning (1956). The specific purpose of this study was to determine the effects of fire on the reduction of shrub species and the subsequent

production of herbaceous plants. Burroweed was reduced by nine-tenths, cholla by about one-half, pricklypear by one-fourth, and mesquite only by one-tenth by a single burn. The production of all perennial grasses, including Rothrock grama, Arizona cottontop, and Santa Rita three-awn, also was reduced by the burning treatments; but, most of these forage species recovered by the second growing season after the fire and, for some forage species, increased by the fourth growing season in comparison to adjacent unburned areas. However, black grama, one of the more important forage species on the range, was damaged seriously by burning and did not recover during the 4 year study period.

An accidental fire in a desert shrub and grassland community on the Santa Rita Experimental Range provided an opportunity to obtain additional information on the first-year effects of fire on mesquite, Lehmann lovegrass, and black grama (Cable 1965). The fire killed 25% of the mesquite trees on an area with a Lehmann lovegrass cover, compared to 8% on an area with black grama. These differences were attributed to the greater accumulations of herbaceous fuels in the area dominated by Lehmann lovegrass. Less than 2% of the original Lehmann lovegrass plants and only 10% of the black grama sprouted immediately following the fire. However, many "new" Lehmann lovegrass seedlings became established on both burned areas in the first growing season after the fire.

The effects of fire in another desert shrub and grassland community on the Santa Rita Experimental Range was studied by Cable (1967) to determine the relative impacts of planned burnings on the control of shrub species and the production of perennial grasses. Results obtained from this work showed that fire was ineffective in controlling mesquite and only "fair" against cactus. Burroweed was killed easily by fire, but it came back quickly with adequate winter precipitation. The production of perennial grasses, including Santa Rita three-awn, Rothrock grama, Arizona cottontop, and tanglehead, was the same on both burned and unburned areas in the first growing season after the burns. However, production was lower on the burned areas in subsequent years, which was attributed largely to the "heavier" grazing by livestock on these areas.

Two fires caused by human activities (but not set intentionally) in desert shrub and grassland communities on the Research Ranch, near Elgin, Arizona, were subsequently evaluated in terms of the response of the affected vegetation to these fires (Bock et al. 1976). Measurements of the production of herbaceous plants were not taken, although measures of vegetative compositions were obtained and can provide an insight into the effects of the fires. The perennial grass cover was reduced initially by the fires, while the growth of forbs was encouraged. After the second growing season following the fires, the vegetative compositions on the burned areas were 90% similar to those on the adjacent unburned areas. It was concluded, therefore, that the vegetation did not experience any permanent "alterations" as a result of these fires.

In his study of the longer term effects of fire on shrub and perennial grass species, conducted in desert shrub and grassland communities in the Guadalupe Mountains of eastern New Mexico, Ahlstrand (1982) reported that the shrub mortality (considering all species collectively) generally was low after the fires, and most of the shrub species responded vegetatively to burning by sprouting. However, since the annual growth increments of these shrubs contributed little to the total biomass, years would be required for many of these species to obtain prefire stature. The coverage of all perennial grass species, including black grama, plains lovegrass, wolftail, and blue grama, had increased (or, at least recovered) on the burned sites 3 years after the fire. From the results of this study, it was suggested that periodic fires can be utilized to maintain and even increase the occurrence of perennial grasses at the expense of shrubs in this vegetative community.

Improvement of Habitats for Wildlife Species

The habitats for bird populations in the desert shrub and grassland communities of the southwestern United States occasionally are improved by fire. This potential value of fire can be illustrated by the study of the two fires on the Research Ranch (Bock et al. 1976).

On the Research Ranch, total bird numbers were significantly higher on the burned areas than on the unburned areas, due largely to an increase in two common species, mourning dove and chipping sparrow (Bock et al. 1976). In addition, 13 of 19 bird populations observed after the fires were larger in numbers on the burned than on the unburned areas. This positive response to fire probably was because of the floristic changes after the fires, resulting in an increase in the diversity and productivity of the birds.

Fire also can improve the habitats for deer in desert shrub and grassland communities, depending upon the nature of the burn (Kittams 1972). Fires, and particularly recurring fires, that burn in an irregular pattern, leaving unburned patches, often create a diversity of habitat conditions.

CONCERNS AND CONSTRAINTS

From the above, it is apparent that fire in southwestern ecosystems is frequently beneficial in terms of the effects on natural resources. However, to be effective in meeting specified objectives, the use of fire should be considered only on a "case by case" basis. In addition, while the

opportunities for fire management can be good on many sites, it is only fair to point out the fact that there are a number of concerns and possible constraints that must be considered before burning programs become operational. Some of the concerns and constraints in southwestern ecosystems are described briefly in the following paragraphs.

Environmental Factors

The use of fire can be constrained by smoke management consideration, and the number of areas constrained likely will increase in the future because of the air quality legislation currently being implemented (Sandberg et al. 1979). Unless the smoke management systems are provided a better technical base, objectives that can be met most effectively (and often, only) with fire may not be satisfied. Smoke management requires the application of technology to minimize the impairment of air quality by burning. The practice of smoke management includes both fuel management and fire prescriptions that improve combustion efficiencies, firing and subsequent "mop up" techniques to reduce emissions, scheduling to enhance convection and dispersion, and scheduling to ensure plume trajectories away from "sensitive" areas. In essence, the central principle of smoke management is to avoid an overloading of natural clearance mechanisms.

That fire affects the nutrient cycles in southwestern ecosystems is known (Klemmedson 1976, Covington and Sackett 1984, DeBano et al. 1987). The use of fire on sites with fuel accumulation of many years can result in substantial changes in surface organic materials and nutrient storages. In a study in a ponderosa pine forest near the Fort Valley Experimental Forest, for example, Covington and Sackett (1984) reported that the storages of surface organic materials were reduced greatly by a prescribed fire, with nutrient storages somewhat less affected. Significantly, the burning released much of the nutrients bound in the surface ogranic materials, improving the conditions for plant growth. Nevertheless, as pointed out by Klemmedson (1976), the guidelines for prescribed burning seldom fully recognize the impacts of fire on nutrient regimes or include the potential loss of nutrients as a consideration in fire planning.

It also is important to understand the effects of fire on soil. Fire generally destroys the organic materials on a soil surface, or residues that eventually become organic materials (Wells et al. 1979). Burning of the organic materials on the soil surface removes or decreases the protective litter layers, volatilizes large amounts of nitrogen and smaller amounts of other elements, and transforms less volatile elements into soluble forms that are more easily absorbed by plants or lost by leaching. Heating of the underlying soil layers also alters the physical, chemical, and biological properties of the soil that are dependent upon the organic materials on the soil surface. These general relationships lead one to think that the effects of fire on soil are predictable. However, to the contrary, these effects are variable, as suggested by studies of these relationships in southwestern ecosystems (Mayland 1967, Wagle and Kitchen 1972, Wagle and Eakle 1979, Covington and Sackett 1986, Kovacic et al. 1986, Ryan and Covington 1986). Information from these and other studies can provide a basis for evaluating ecosystem alterations in fire management.

Water repellency of soil caused by fire has received considerable attention in recent years (DeBano 1981). All mineral soils containing only small amounts of organic materials are likely to become water repellent, to some degree, when heated. The severity and distribution of the water repellency after a fire will determine the subsequent land management problems on the site. Fortunately, prescribed fires afford a practical method of modifying this water repellency by controlling the occurrence and behavoir of the fire, as opposed to permitting the fuels to accumulate and burn by wildfire.

Economic Factors

In a detailed discussion on the analysis of fire management programs for economic efficiencies, Althaus and Mills (1982) indicated that the same basic valuation principles that apply to an economic efficiency analysis of any natural resource program also apply to fire management programs. A fire analysis problem, however, includes some unique requirements for both the structure of the analysis and the application of the valuation principles. Nevertheless, the most appropriate economic selection criteria among alternative fire management programs (including the option of no fire) is the minimization of the fire management cost, plus the net change in natural resource outputs, that is, C + NVC (Althaus and Mills 1982). The C + NVC criterion recognizes both the beneficial and the detrimental effects of fire. To the extent possible, therefore, source data sets which are site-specific must be available to adequately quantify costs and changes in the natural resource values for the alternative fire management programs considered. Unfortunately, these data sets are not complete for all situations in southwestern ecosystems.

Information on the costs of burning in southwestern ecosystems is scarce. This lack of information is attributed largely to previous deficiencies in reporting procedures (Wood 1988). Examples of cost data that are available represent broadcast burning (Wood 1988), and piling and burning of slash (Turner and Larson 1974) in ponderosa pine forests. To properly evaluate the economic effi-

ciencies of alternative fire management programs, it is obvious that additional cost information representing a range of conditions and sites is necessary.

Changes in natural resource values caused by fire should consider both the effects of fire on values at the fire site and the effects away from the site, a valuation "rule" not adhered to in all instances in southwestern ecosystems. It also was pointed out by Althaus and Mills (1982) that all natural resource values that can be valued in monetary terms should be included in the calculations of net value change; the values that cannot be measured readily in monetary terms should not be "forced" into an analysis of economic efficiencies. It is important to evaluate the changes in natural resource values, both losses and benefits, in the long-run, as these values can change drastically over a period of several years after a fire (Lowe et al. 1978). Such a consideration has not necessarily been the case in southwestern ecosystems.

Educational Factors

In the "final" analysis, the opportunities for fire management in southwestern ecosystems will depend largely upon public support. That the public recognizes that fire can be both beneficial and detrimental has been shown by Cortner et al. (1984), who also determined that the public acceptance and understanding of the purposes and benefits of fire management are high, in relative terms. Furthermore, additional knowledge of the effects of fire on natural resources can increase the "tolerance" for fire by the public (Taylor et al. 1986). Finally, to be most effective in obtaining the necessary public support for fire management programs in the future, education should be oriented to local conditions in the southwestern ecosystems to be affected, as well as to local knowledge and acceptance of fire management.

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Fire History in a Small Ponderosa Pine Stand Surrounded by Chaparral[1]

J. H. Dieterich and A. R. Hibbert[2]

Abstract.—Before intensive mining activity began in the 1860's, fires burned at an average 2-year interval within a 215-acre ponderosa pine *(Pinus ponderosa)* stand surrounded by chaparral in the Bradshaw Mountains in central Arizona. It is postulated that in the vicinity of the study area a natural fire-induced mosaic of uneven-aged chaparral developed in the pre-mining period because of the frequent ignition source in the adjacent pine stand. The ponderosa pine and associated oak and juniper trees were heavily cut during the expansion of local mining activity from 1863 to 1885, after which fire protection, low fuel loading, and grazing essentially eliminated large fires for many years. Today, most of the chaparral stands are overmature, lack a natural mosaic appearance and contain heavy accumulations of dead material.

Located on the Prescott National Forest, 20 miles south of Prescott, Arizona, the Battle Flat pilot watershed was established in 1978 to evaluate the various techniques available for manipulating chaparral cover to enhance water yield and other resources. The predominantly shrub cover is typical of thousands of acres of chaparral in central Arizona. A stand of ponderosa pine situated in the lower reaches of the watershed has been developing and maturing over the past 100 years (fig. 1). Severely weathered stumps provided evidence that this same general area has been occupied by pine for at least 300 years. The pine stand, including oak and juniper trees, was cutover in the late 1800's to provide wood products for the mining activity that began in the 1860's. If prescribed burning is to be used as a technique for managing or manipulating chaparral, a general understanding of the history of fire in the area would be useful.

Although no historical fire frequency data are available for Arizona chaparral, fire suppression records indicate that fires have been common. Lightning provides a more-or-less uniform fire risk, and results in a relatively consistent source of ignitions. Fires may have been set intentionally by Native Americans, although there is little reliable evidence to either confirm or refute this theory. Prospecting and mining activity undoubtedly caused fires in the area, but fires occurring before the 1860's could not logically be attributed to this cause.

Chaparral is well adapted to fire; however, when chaparral burns, the fire hazard is modified for many years. Most chaparral shrubs are prolific crown and/or root sprouters that live to considerable age. Pond (1971) found little change in species composition after 47 years, with the

Figure 1.—Lower central Battle Flat watershed looking south. The oldest trees in the ponderosa pine stand, which are mostly visible here, were aged at approximately 100 years.

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Former Research Forester and Former Research Hydrologist, Rocky Mountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287-1304.*

exception that there was an increase in the amount of dead biomass within the plants.

Fire behavior and fire spread in chaparral are dependent upon a number of weather-related factors (Davis and Dieterich 1976). Total amount of available fuel, and the ratio of living-to-dead fuel material are also important. Although there is no documented research on the subject, those experienced in using fire and suppressing fires in chaparral generally agree that if the chaparral stand is less than 10 years old, fire spread is unlikely (unless a grass cover has invaded). Dead material begins to accumulate in stands 10-15 years old, and although in these stands fires may spread upslope or with the wind, sustained spread is not likely. By about the 20th year, the dead-to-living fuel ratio has increased to a level where continuous fire spread may be expected. Stands older than 20 years can only become more flammable as the amount of dead material in the stand increases.

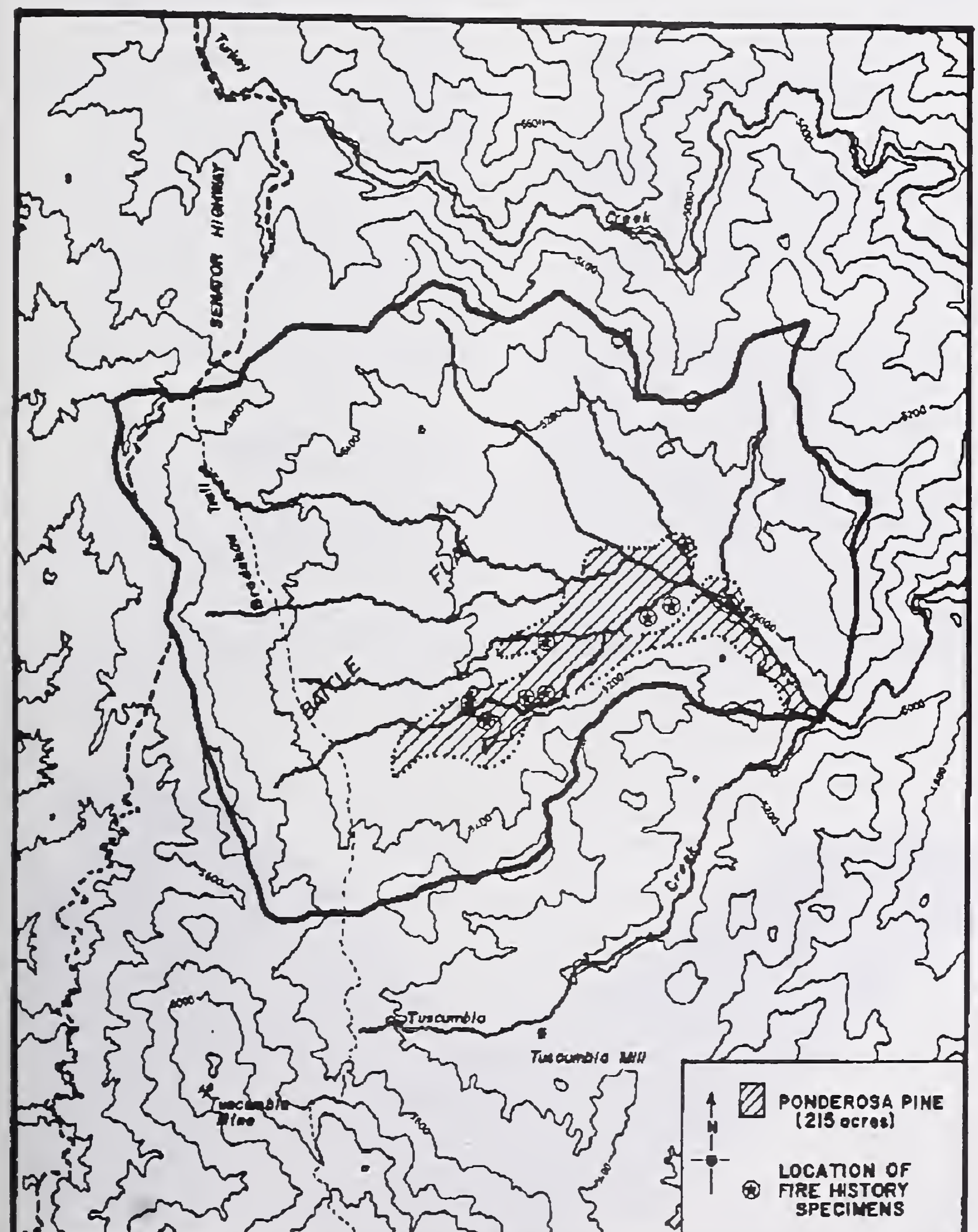

Figure 2. The Battle Flat study area is at an elevation of 5,000 to 6,000 feet along the crest of the Bradshaw Mountains, 20 miles south of Prescott, Arizona.

Documenting fire history for a particular chaparral stand is difficult because, for the most part, old fire-scarred material is not available that can be accurately dated with standard techniques of dendrochronology. The Battle Flat area offers a unique opportunity in that a reliable record exists of historical fires burning in a ponderosa pine stand surrounded by chaparral. Fire history information from this stand of ponderosa pine was used to determine the fire regime in the ponderosa pine, and assisted in formulating a theory on the role that fire has played in development of the chaparral stand in the area.

Stand Description and Data Collection

The Battle Flat study watershed and the area of the existing ponderosa pine stand are shown in figure 2. At an elevation of 5,000 to 5,300 ft and encompassing approximately 215 acres, the stand is best described as uneven-aged with a scattered association of alligator juniper *(Juniperus deppeana)* and Arizona white oak *(Quercus arizonica)*. The habitat type is ponderosa pine/Arizona white oak *(Pinus ponderosa/Quercus arizonica)* (Muldavin et al. 1986). Canopy cover of coniferous trees is about 45%. There are few trees that might be described as fully mature (yellow pine), but most of the large trees still fall in the immature (blackjack) category. Regeneration has been adequate and there appears to be a good distribution of age classes (not measured). The dominant vegetation on the periphery of the pine stand is shrub live oak-mountainmahogany *(Quercus turbinella-Cercocarpus breviflorus)* and shrub live oak-mixed shrub plant communities. Beyond the irregular perimeter of the pine stand, scattered young pine trees have become established in the chaparral; thus, it ap-

pears that the pine stand has been slowly encroaching into the chaparral. The extent to which this can be expected to continue is unknown.

The seven fire history specimens used for this analysis came from the ponderosa pine stand located along or near stream channels in the lower portion of the watershed (fig. 2). All samples were taken from dead material—mostly stumps that exhibited fire scars on the base, or from pitch-impregnated wood fragments that remained intact after decay had destroyed most of the stump. Despite a careful search, no living trees were found with fire scars, nor was there any other recent evidence of fire in the stand. Increment cores were also taken from the largest and oldest appearing trees within the area to determine their age.

Analysis of Material

Seven specimens were collected and prepared for analysis and dated by dendrochronologists at the Laboratory of Tree-Ring Research, University of Arizona, Tucson. The specimens were very fragile and an epoxy was used to stabilize them before cuts were made across the grain to expose the ring pattern (fig. 3). The cross sections were smoothed with a sander to improve ring visibility and dating accuracy. Cross-dating, a standard ring-dating procedure used in dendrochronology (Stokes and Smiley 1968), was used to date all rings on the specimens. Then, once the fire scars were identified, the dated ring in which each scar occurred was determined (Dieterich and Swetnam 1984). The original dating and fire scar identification done by one person was verified independently by a second dendrochonologist. Any disagreements on scars or dates were rechecked and differences resolved to arrive at the final dates.

Because of the advanced stages of decay in the wood surrounding the fire scars, many of the specimens lacked a pith, date (center of the tree was not included), and/or a cambium (the last ring produced prior to cutting or death of the tree). However, cross-dating made it possible to precisely date each ring even when these years were unknown. The fire dates were then plotted on a chart to become the Master Fire Chronology for the area.

Dating of the increment cores failed to locate any trees older than 100 years.

History of the Area

A brief review of historical events and activities in the area may help explain some of the vegetative changes that have occurred in the past, and conditions as they exist today.

Gold was discovered in the Bradshaw Mountains in 1863 and placer mining began soon after on streams such as Turkey Creek. However, the remoteness of the area and the hostility of the resident Apache Indians caused mining developments to proceed slowly. The mining camps of Goodwin, 5 miles north of Battle Flat and Bradshaw City to the south, became active in the late 1860's and early 1870's (Jaggar and Palache 1905). The discovery of silver deposits near Battle Flat spurred new mining activity in the 1870's.

The Peck Mine, 4 miles southwest of Battle Flat, started in 1875 and produced a million dollars or more in the first 5 years. Other nearby mines produced large quantities of silver, but the rapid exhaustion of the rich ore and the fall of the price of silver brought this period to a close by 1885.

The Tescumbia Mine and Mill appear to be the mining operations closest to Battle Flat (fig. 2), although the Peck Mine also may have utilized timber products from the area. It seems certain that the pine-oak-juniper stands found in Battle Flat were heavily utilized for fuelwood, mine timbers, and general construction. In fact, it appears that the area was largely cutover and most of the usable material removed by about 1880.

Results

The Master Fire Chronology (fig. 4) provides a summary of the length of record for each specimen, the individual years in which fires were recorded on each specimen, gaps in the record or periods when information

Figure 3.—Specimen 1 was collected in April 1979 and typifies the material used in the fire history study. The fire-scarred portion of the cross section was preserved over the years by heavy pitch; the remainder of the stem rotted away.

appears to be missing, and the total number of fire years recorded by the available cross sections.

Detailed inspection of the Master Fire Chronology (MFC) indicated the following:

1. The MFC covers the period 1700-1874 (174 years).

2. The oldest dated ring was 1653.

3. The oldest fire scar was recorded in 1700.

4. Specimen No. 4 recorded 25 fires during the 93-year period 1781-1874, for an average interval of 1 fire every 3.7 years. Specimen No. 4 also recorded the longest period between fire scars, 45 years (1736-1781).

5. The longest continuous period of scarring was recorded on Specimen No. 6. During the 149-year period (1700-1849), 25 fires were recorded, for a mean fire interval of 6 years.

6. Perhaps the most environmentally significant data comes from the composite record of fires for the 100-year period (1770-1870). During this period all seven specimens were contributing about equally to the composite. Fires occurred somewhere within the 215-acre area in 67 of the 100 years, for an average fire interval of 1 fire every 1-2 years (1.5-yr interval).

7. If all the available data were used, starting with the first recorded fire in 1700 and ending with the last fire in 1874, we find 92 fire-years in the 174-year period for an average fire interval of about 1 fire every 2 years (1.89 yr).

8. The Master Fire Chronology provides a record of several periods when fires occurred somewhere within the pine stand in consecutive years. For example, fires were recorded to have burned somewhere within the stand in 9 consecutive years (1782-1790); 5 consecutive years (1848-1852); and 4 consecutive years (1812-1815, 1817-1820, and 1822-1825).

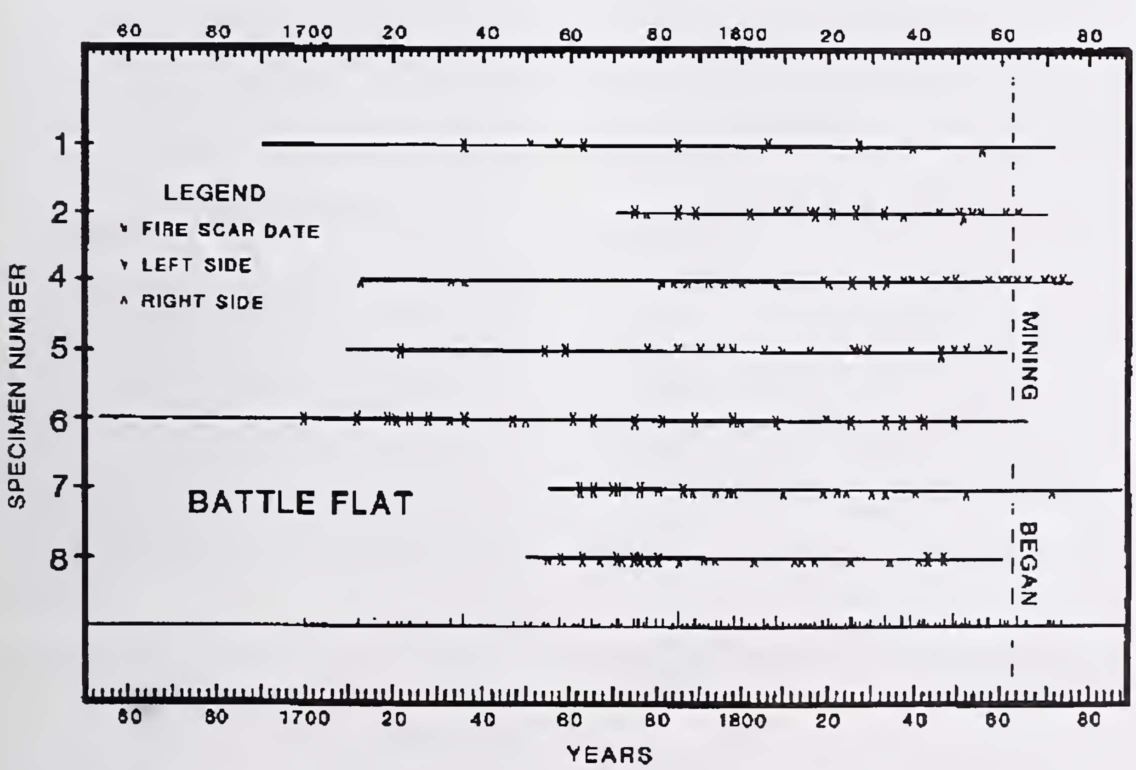

Figure 4. Master Fire Chronology for 7 specimens collected in Battle Flat. The lower line contains the composite record of fire scars for all specimens.

No fire scars were discovered on living trees (all living trees were less than 100 years old), and there is no evidence of recent fires in the area. A logical question might be, "Why, after such an extended period of fire activity in the area, should there be another extended period when there were no fires?" The absence of recent fires in the area agrees with the results of two fire history studies in Arizona on the Coconino National Forest adjacent to the Prescott National Forest. At Chimney Spring on the Ft. Valley Experimental Forest, north of Flagstaff, Arizona, the last recorded fire was in 1876 (Dieterich 1980a). At Limestone Flats, at an elevation of 6,900 ft but only 65 air miles to the east of the Battle Flat area, the frequency of fire essentially terminated at the turn of the century, with the last general fire on the area in 1898 (Dieterich 1980b, Dieterich and Swetnam 1984). Probable explanations for this decrease in fire incidence include the reduction in use of the area by Native Americans, more intensive fire protection, change in characteristics or condition of the pine stand and/or the surrounding stands of chaparral, and a change in occurrence of lightning-caused fires.

Lightning occurrence probably has not changed much over the years; but the characteristics of the pine and chaparral stands have changed and, at least in the pine stand, this may have contributed to the change in fire incidence. The stand of ponderosa pine that was present before 1863 when the mining began was in marked contrast to the present stand. The pre-1863 stand would have included a range of size and age classes, including many old-growth trees. In fact, Specimen No. 6 was at least 300 years old when it was cut in 1863. There would have been an

abundance of standing snags and, although surface fires were frequent, there would have been a sufficient amount of dead and rotten material available in which lightning fires could propagate. Some large down and dead material would have been present, the ground fuels would have been predominantly needles, grasses, and forbs; these light fuels would have been present in sufficient quantities to support the spread of fire.

The current stand represents the early stages of development in ponderosa pine succession. Few snags are present and heavy ground fuels are absent, reducing the chances for lightning to start fires in rotten logs. Pine needles are the predominant ground cover; grasses and forbs are sparse, are continually grazed, and cannot compete with the heavy mat of needles that continues to build.

Management Implications and Conclusions

Fire history information from the ponderosa pine stand in the Battle Flat area provides direct evidence of the influence that fire has had in the pine stand, and indirect evidence of the role that fire has played in the development of the area's chaparral stands.

Because of land-use changes that were occurring around the turn of the century, it is helpful to look at the fire history during three separate periods (pre-mining, mining, post-mining) and determine some of the implications for present and future management of the area.

Pre-mining Fire History (Before 1863)

The fire history data show that fires occurred frequently in the Battle Flat pine stand before 1863 when the first mining began in the Bradshaw Mountains. Lightning has always been a source of ignitions, and Native Americans, in the area at the time, probably contributed to fire starts.

With fires burning on an average of 2-year intervals, a fairly open stand of pine would result with a ground covering of grasses and forbs. Fuels needed to sustain ignition and spread of fire would come from this herbaceous material and from the annual needle-fall from the pine overstory. Short fire intervals would result in fires of very low intensity whose behavior would range from a very rapid spread in dry, windy conditions, to a very slow patchy spread during less favorable burning conditions.

We can infer that many of the fires starting in the ponderosa pine probably reached the perimeter of the stand where pine trees mixed with the shrubs. If the chaparral stand was less than 10 years old and the grasses had not come in, fire spread would be stopped. If the chaparral had gone 15-20 years without a fire and/or if there was a grass cover, the fire might continue to spread into the chaparral. The chaparral stands that existed during the pre-mining period could logically be characterized as being a "mosaic" in which brush stands of all ages (up to about 20-25 years) would have been represented. With this type of mosaic, large fires like the Battle Fire in 1972 that started on the edge of Battle Flat and burned more than 28,000 acres would have been a rare occurrence.

The fire history in the Battle Flat area during the pre-mining period was characterized by fires burning at very short intervals in the pine, and fires burning in the chaparral at intervals of not less than 15-20 years.

Active Mining Period Fire History (1863-1885)

Fire history material is unavailable for the period beginning soon after the start of placer mining in the Bradshaw Mountains (1863), and only Specimen No. 4 recorded fires after 1871 (1873 and 1874), the year when the major silver mining activity began.

Since most of the stands had been cut by 1874, we can only speculate on what the fire occurrence was like during this period. Lightning strikes remained a normal occurrence, and there may have been land-clearing fires designed to open up the country to improve access and movement. Thus, the burned mosaics in the chaparral stands were possibly still maintained because of the increased settlement in the area and the fact that there was probably little effort to control fires except when they threatened mines or improvements. Grazing may have also begun to interupt the fire cycle.

Post-mining Period Fire History (1885 to Present)

There was only scattered mining activity in the area after the turn of the century. However, grazing continued and, with the advent of fire protection, aided by the existing natural mosaic pattern, and by the fact that frequent fires in the late 1800's had prevented accumulations of fuel, early fire protection efforts would have been largely successful in preventing large fires.

Improved protection (in the absence of prescribed burning) resulted in a progressive increase in areas of mature and overmature stands of chaparral. By 1920, most of the effects from the natural mosaic burning common until the 1860's had disappeared and large continuous areas of chaparral were developing that were at least 20 years old, with many much older.

Today, thousands of acres of chaparral have long since passed the mature stage and exist in an overmature condition. The Battle Fire that started along the western edge of Battle Flat (but did not burn the watershed)

started in brush cover that had probably been growing without interruption since around the turn of the century. Although an aggressive fire protection program has been largely successful in extinguishing chaparral fires when they are small, the fire hazard in the chaparral stands is actually increasing. Fuels are present in sufficient quantities and over such large continuous areas that large fires in the future are inevitable.

In conclusion, land managers responsible for protecting and managing these areas must recognize that (1) large fires will continue to occur at indeterminant intervals, and under conditions that are dictated largely by weather, ignition sources, and available suppression forces; (2) although prescribed burning is a useful, environmentally acceptable tool, it is unlikely that it can be practiced on a scale sufficient to restore the natural mosaic patterns present in the chaparral stands at the turn of the century; and (3) because of (1) and (2) above, the manager must carefully select the areas to be treated with prescribed fire, basing decisions on the resource needs and on the strategic location of the area in relation to regions of high fire risk requiring special protection measures.

Acknowledgement

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Effects of Tebuthiuron and Fire on Pinyon-Juniper Woodlands in Southcentral New Mexico[1]

Roger D. Wittie and Kirk C. McDaniel[2]

Abstract.—Spring burning of pinyon-oneseed juniper woodlands previously treated with tebuthiuron was evaluated to determine its effects on trees, associated shrubs and herbaceous vegetation. Pinyon pine *(Pinus edulis)* was more affected by herbicide and less affected by fire than was oneseed juniper *(Juniperus monosperma)*. Canopy reductions were dependent on tree density and range site. Forage production increased after herbicide and burning treatments whereas browse production declined. Seedling densities for trees were consistent across sites but affected by herbicide and fire treatments.

Pinyon-juniper (P/J) woodlands are one of the most extensive vegetation types in the southwestern United States. Springfield (1976) estimates 3.6 million ha of P/J in New Mexico with about 160 million ha in the southwestern states (Aro 1971). Historical evidence suggests P/J woodlands have increased in range and tree density since the turn of the century. Migration into perennial grasslands along the woodlands' lower elevational range has decreased forage production for livestock, increased poisonous plant densities, created soil erosion problems and increased the costs of livestock management (Parker 1948, Johnsen 1962 and Smeins 1983).

Herbicide treatments generally have not been used widely to reduce P/J dominance (McDaniel and WhiteTrifaro 1987). Control of pinyon-oneseed juniper by herbicides is highly dependent upon both range site and tree growth characteristics (McDaniel et al. 1989). Prescribed burning has also not been widely used in P/J woodlands because results are often inconsistent. Integrated (complementary) brush management systems using two or more control methods may be needed in order to meet multiple use objectives. Scifres (1987) has shown that application of a herbicide followed in later years by a controlled burning program can offer an efficient way to manage brush canopies when long-term economics are considered. The primary objective of this research was to examine the response of trees and shrubs following application of the herbicide tebuthiuron (*N*-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-*N*,*N*-dimethylurea) and/or fire. A secondary objective was to monitor change in herbaceous composition and yield following treatment.

Methods

This study was conducted at the New Mexico State University Ft. Stanton Experimental Ranch in Lincoln County near Capitan, New Mexico. Soils and vegetation have been previously classified (Groce and Pieper 1967, Pieper et al. 1971). Average annual precipitation is 390 mm with about 250 mm falling from May to September. Within the study area, three range sites (hill, shallow and loamy) were identified. Each range site has a different slope, soil, tree density and mixture of herbaceous vegetation. All sampling was stratified by range sites within treatment blocks.

Hill range sites were characterized by 10% to 45% slope, a maximum soil depth of 30 cm and tree densities of about 1900 individuals/ha. Soils were predominantly lithic haplustolls classified as Tortugas-rock outcrop. Herbaceous vegetation was dominated by sideoats grama *(Bouteloua curtipendula)*, New Mexico muhly *(Muhlenbergia pauciflora)* and plains lovegrass *(Eragrostis erosa)*. Shallow range sites were characterized by slopes of 3% to 10%, maximum soil depths of 100 cm and tree densites of about 700 individuals/ha. Soils are predominantly petrocalciustolls classified as Plack variant gravelly loams. Herbaceous vegetation is dominated by sideoats grama, blue grama *(Bouteloua gracilis)*, wolftail *(Lycurus pheloides)* and various threeawns (*Aristida* spp.). Loamy range sites are characterized by slopes of 1% to 10%, maximum soil depths over 150 cm and tree densities up to 235 individuals/ha. Soils were predominantly aridic haplustolls classified as Deacon loams. Tree cover was almost solely afforded by oneseed juniper. Herbaceous vegetation was dominated by blue grama, galleta *(Hilaria jamesii)*, creeping muhly *(Muhlenbergia repens)* and sand dropseed *(Sporobolus cryptandrus)*.

Tebuthiuron pellets (40% active ingredient) were applied to P/J woodlands within the study area in August 1983. Rates of 0.84, 1.26 and 1.40 kg/ha active ingredient were

[1] *Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2] *Wittie is graduate research assistant, and McDaniel is Professor, Department of Animal and Range Sciences, New Mexico State University, Las Cruces, NM 88003.*

flown onto 6 ha (1250 X 120 m) blocks placed side by side, each separated by a 35 m buffer area. One-third of each herbicide block, the buffers and a small control area were burned in March 1987 using head-fire/backfire techniques (Wright 1974 and Scifres 1980). Fire containment was accomplished by burning out fuels around the perimeter of treatment areas. Fuel moisture and fire weather conditions were monitored for 2 weeks prior to actual ignition.

One-tenth ha sampling clusters were established on each range site within treatment blocks. Five trees of each species were randomly selected from each cluster for evaluation. Evaluations included tree height, canopy cover, tree position (overstory/understory), percentage canopy removed by herbicide and percentage canopy damaged by fire. Herbaceous sampling was conducted using 0.186 m^2 rectangular frames. Frames were placed beneath the dripline of sample trees and 1 m from the dripline into the interspace area on the tree's east/west axis producing four quadrats per tree and 40 per cluster. Annual production values by species were generated using weight-estimate techniques (Cook and Stubbendieck 1986). Each species encountered within four quadrats was clipped and weighed to form a green weight regression between estimated and actual values. Clipped samples were oven-dried at 60° C for 72 hours and weights were adjusted to dry forage values. Fire damage and tree/shrub seedling densities were estimated using ten 7.18-m diameter subplots within each cluster. All trees greater than 0.3 m in height within a subplot were counted and evaluated prior to, and at 6 and 12 months after burning. Seedling densities were taken from within subplots at the same intervals. All data were analyzed in a randomized complete block design and subjected to mean separation using protected LSD, Duncan's multiple range test or Chi-squared analyses (Steele and Torrie 1980). Because treatments were applied to only one block, pseudo-replication (over time) was used to estimate experimental error for this analysis. Therefore, extrapolation of differences to other areas must be done with caution and values reported here are preliminary results.

Results

Tree and Herbage Response to Tebuthiuron

Pinyon was highly susceptible to tebuthiuron regardless of rate or range site (table 1). Mortality averaged 60%, 58%, and 78% from the low to high rate, respectively. Oneseed juniper was less susceptible to the herbicide at low and moderate rates (mortality was 10% and 43%), compared to the high application rate (74%). Generally, moderately large pinyon (those 2.5 to 4 m in height) and smaller juniper (those less than 2.5 m in height) were most susceptible to the herbicide (McDaniel et al. 1989).

Grass yield did not increase in the herbicide treated areas compared to the untreated areas until the second or third growing season after treatment (table 2). After the third year, average grass yield increased with increasing rate of application, being 615, 775 and 1000 kg/ha compared to 428 kg/ha for the untreated area. Initial depression and subsequent release of grass yields after 2 to 3 years is common for soil applied herbicides (Duncan and Scifres 1983 and James and Pettit 1984). Forb production decreased uniformly across herbicide rate the first and second growing season after treatment. By the third season, forb production began to increase in areas treated with the low and medium rate of herbicide but remained depressed in areas treated at the high rate. Several studies found forbs to recover between 18 and 36 months after treatment with tebuthiuron depending upon applied rates (Scifres and Mutz 1978 and Jacoby et al. 1982). Perennial grasses comprised about 70% of the composition by weight in untreated areas but nearly 95% in areas treated at the high tebuthiuron rate after three years. Total herbage yield also varied by range site within a treatment area. Overall, herbage yields averaged 810, 915 and 1334 kg/ha on shallow, hill and loamy sites, respectively. Peak yields from tebuthiuron treatment may not yet be realized. Arnold (1964) reported peak herbage yields following tree removal in pinyon-juniper woodlands of Arizona occurred 12 years after treatment. Pro-

Table 1.—Percentage mortality of pinyon and oneseed juniper after aerial applications of pelleted tebuthiuron on the Ft. Stanton Experimental Ranch near Capitan, NM in September 1987.[1]

		Percent mortality by tebuthiuron rate (kg/ha)		
Species	Range site[2]	0.84	1.26	1.40
	Hill	59^{b}	54^{b}	81^{a}
Pinyon	Shallow	61^{b}	59^{b}	75^{a}
	Loamy	-	-	-
Hill		8^{c}	31^{b}	77^{a}
Juniper	Shallow	3^{c}	27^{b}	75^{a}
	Loamy	19^{c}	17^{b}	70^{a}

[1]Row means followed by different superscripts are significant ($P<0.05$) according to Duncan's multiple range test.

[2]Too few pinyon occurred on loamy range site to achieve an adequate sample.

duction values given in table 2 for 1986 represent the quantity of fine fuels present in the study area prior to burning.

Tree and Herbage Response to Burning

Table 3 indicates canopy reductions achieved from burning treatments applied 4 years after tebuthiuron. Sixty percent of the remaining juniper in tebuthiuron treated areas were damaged. Fire was effective in reducing oneseed juniper canopies within areas previously treated with tebuthiuron at the low and medium rate but did not further reduce canopies in areas treated at the high tebuthiuron rate. Oneseed junipers were damaged slightly (5% to 20% crown scorch) on hill and shallow sites where fuel beds were light or inconsistent but heavily damaged where adequate fuels were present. Oneseed juniper on loamy range sites received moderate to heavy damage (25% to 75% crown scorch) at the low and medium herbicide rate. Those trees less than 2 m in height were heavy damaged or removed by fire. Larger juniper received light to moderate crown scorch and generally where crowns were in close proximity, complete crown fires developed. Similar results were reported after burning herbicide-treated eastern redcedar *(Juniperus virginiana)* in Oklahoma (Engle et al. 1988).

Few pinyon were damaged by fire where burning was not preceded with tebuthiuron application. Where tebuthiuron had been applied, fuel loading and distribution were improved. Twenty percent of the remaining pinyon were damaged by fire. Most exhibited scorching around the bole and 15% of the canopies were slightly scorched (table 3). Smaller pinyon (those less than 1.2 m in height) received the heaviest damage to the canopy or were burned out completely. Larger pinyon (greater than 1.3 m in height) generally survived burning except where oneseed juniper canopies enclosed the pinyon's and crown fires developed. The relationship between tree height and percentage of canopy removed by fire has been reported from other studies (Dwyer and Pieper 1967 and Engle et al. 1988).

The tebuthiuron-fire treatment increased herbage production above that of either treatment alone. By further reducing tree canopy cover at

Table 2.—Annual herbage yield (kg/ha) for tebuthiuron and tebuthiuron plus burning treatments in pinyon-oneseed juniper woodlands at Ft. Stanton.[1]

		Tebuthiuron rate[2] (kg/ha active ingredient)							
	Growing season precipitation[4]	-0-		0.84		1.26		1.40	
Year[3]	(mm)	Grass	Forb	Grass	Forb	Grass	Forb	Grass	Forb
1984	244	432^{a}	96^{a}	462^{a}	9^{b}	416^{a}	21^{b}	418^{a}	10^{b}
1985	201	337^{b}	84^{a}	478^{a}	15^{a}	673^{a}	22^{a}	667^{a}	40^{a}
1986	323	428^{c}	112^{a}	615^{bc}	42^{b}	775^{ab}	28b	1000^{a}	70^{ab}
1987a	778^{b}	185^{a}	654^{a}	101^{c}	600^{a}	89^{bc}	1373^{c}	46^{b}	
1987b	-	-	-	710^{a}	155^{a}	730^{b}	214^{a}	1302^{a}	55^{a}

[1]Row means for plant components followed by different superscripts are significantly different ($P<.05$) by Duncan's multiple range test.

[2]Tebuthiuron (40% active ingredient) pellets aerially applied in August 1983.

[3]Years 1984 through 1987a represent means pooled across range sites following tebuthiuron application only. Year 1987b represents tebuthiuron treatment followed by spring burning pooled across range sites.

[4]Growing season is considered to be May 1 through September 30, with the long-term average of approximately 250 mm.

Table 3.—Percentage crown damage to pinyon and oneseed juniper induced by spring burning applied to range sites treated with tebuthiuron at Ft. Stanton.[1]

			Percent crown damage by tebuthiuron rate (kg/ha)[3]		
Species	Range site[2]	Height class (m)	0.84	1.26	1.40
	Hill	<1.2	4^{b}	8^{b}	7^{b}
		>1.3	0^{b}	6^{b}	0^{b}
Pinyon	Shallow	<1.2	18^{b}	16^{b}	23^{b}
		>1.3	0^{b}	3^{b}	3^{b}
	Loamy	-	-	-	-
	Hill	<1.2	13^{b}	22^{b}	27^{b}
		>1.3	4^{b}	4^{b}	9^{b}
Juniper	Shallow	<1.2	16^{b}	38^{a}	5^{b}
		>1.3	3^{b}	31^{a}	3^{b}
	Loamy	<1.2	43^{c}	62^{b}	8^{a}
		>1.3	21^{c}	44^{b}	8^{a}

[1]Row means followed by different superscripts are significant ($P<0.1$).

[2]Too few pinyon occurred to achieve adequate measure on loamy range sites.

[3]Crown damage scores were assessed at 1 and 6 months after burning and pooled to achieve means.

the low and moderate tebuthiuron rate, burning allowed for an additional release of herbaceous species (table 2). Grass production increased an average of 13%, 19% and 38% across range sites from the low to high herbicide rate, respectively, after burning. Burning stimulated a release of forbs at the low and medium herbicide rate, increasing yields 53% and 140%, but was not effective at the high tebuthiuron rate. Percentage composition by weight for the forb component equalled that of untreated areas after burning the low and medium tebuthiuron rate. In untreated areas annual forbs were a major component of the coppice zones associated with tree canopies whereas perennial forbs were predominant in interspaces zones. The complement of annual forbs increased in both zones after treatment (Arnold 1964 and Arnold et al. 1964).

Regeneration Sampling

As expected, seedling densities of both pinyon and oneseed juniper were decreased immediately after burning (table 4). However, new seedlings began to appear within the first year following fire. Seedling numbers corresponded closely to actual tree densities being greatest on hill range sites and lowest in loamy range sites. Generally, there was little difference in pinyon or oneseed juniper seedling numbers between burning or herbicide with burning treatments. Burning alone appeared to favor oneseed juniper and pinyon establishment on shallow sites, but this may be in part due to a poorly distributed fuel bed across these areas which resulted in a cool or erratic fire front. Seedling numbers for pinyon increased following burning on loamy sites where it only occurs rarely. This suggests the presence of a viable seed source but an inability to become established against a competitive perennial grass component. Research from other P/J woodlands in the Southwest indicates recolonization may take as little as 15 years on some sites after overstory removal (Tausch et al. 1971 and Rippel et al. 1983).

Response of Associated Shrubs

Associated shrubs in these woodlands were susceptible to tebuthiuron rates applied (McDaniel et al. 1989). Wavyleaf oak *(Quercus undulata)* and algerita *(Berberis haematocarpa)* densities were reduced by 80% or more whereas skunkbush sumac *(Rhus trilobata)* was more tolerant (50%) at the high tebuthiuron rate.

Only algerita occurred in high enough numbers on loamy sites to achieve an adequate measure of response in these areas and was heavily damaged or completely removed by fire. Algerita's close association with canopies of oneseed juniper provided it some protection from herbicide but increased its susceptibility to fire. Burning opened microsites for the reestablishment of algerita across all sites. Since this species prefers microsites beneath tree canopies for establishment, once trees are removed, the seed source present is probably released. Oak was more tolerant to burning than was sumac on hill and shallow sites. Wright (1972) found oak sprouts to increase immediately after fire then decrease to preburn levels over 18 years. The combination of tebuthiuron and burning severely depleted cover and browse afforded by both oak and algerita, but stimulated resprouting in stagnant sumac thereby increasing its value as a browse species. Competition from perennial grasses may have been a factor for suppression of these shrubs on loamy sites.

Table 4.—Pinyon and oneseed juniper seedling densities (no./ha) after tebuthiuron and burning treatments applied to three range sites at Ft. Stanton.[1]

		Treatment[2]			
Species	Range site	Control	Tebuthiuron	Burning	Tebuthiuron and burning
	Hill	357^{a}	371^{a}	259^{b}	244^{b}
Pinyon	Shallow	132^{a}	214^{b}	272^{b}	210^{b}
	Loamy	$-0-^{a}$	49^{b}	74^{b}	123^{c}
	Hill	251^{a}	362^{b}	349^{b}	333^{b}
Juniper	Shallow	255^{a}	181^{b}	247^{a}	198^{b}
	Loamy	58^{a}	91^{bc}	124^{c}	74^{ab}

[1]*Row means followed by different superscripts are significant at (P<.05).*

[2]*Means represent data for tebuthiuron applied at 1.40 kg/ha active ingredient only and for a spring burn (March 1987) applied during the 4th season following tebuthiuron application.*

Management Implications

Because of a low understory fuel source, the application of controlled burning under stable, nonhazardous conditions in early spring is generally ineffective in reducing P/J canopies such as those in our study. However, following herbicide application and subsequent herbage release, the application of spring burning holds some promise for further reducing remaining live tree canopies. This may be especially beneficial where herbicide treatments only produced marginal results such as the low and medium rates used at Ft. Stanton. Results from the application of the

medium rate of tebuthiuron achieved a partial canopy reduction and forage release which when later burned produced a mosaic (savanna-like) cover pattern offering both quantity and quality in the forage supplied. Application of a high rate of tebuthiuron alone achieved essentially the same results as the combination of a moderate rate of tebuthiuron followed by burning.

Neither tebuthiuron or burning alone or in combination produced consistent results when seedling numbers of both species were considered. Although treatment can be considered beneficial with respect to some criteria, it also opens microsites for the release of an unquantified seed source. Caution and a complete resource assessment must be utilized, otherwise treatment life may be severely shortened through enhancement of seedling germination. Additionally, above normal precipitation during the 1987 growing season may have been a factor in both herbage and seedling release following treatment. Results may vary following spring burns if soil moisture conditions are unfavorable to plant growth.

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Table 5.—Plant densities (no./ha) of associated shrubs after tebuthiuron and burning treatment on each range site at Ft. Stanton[1]

		Treatment[2]			
Species	Range site	Control	Tebuthiuron	Burning	Tebuthiuron and burning
	Hill	31^{a}	-0-^{a}	24^{c}	44^{a}
Algerita	Shallow	166^{a}	-0-^{b}	74^{b}	-0-^{b}
	Loamy	62^{c}	-0-^{a}	48^{a}	-0-^{a}
	Hill	741^{a}	247^{b}	202^{b}	94^{c}
Oak	Shallow	371^{a}	49^{b}	63^{b}	-0-^{b}
	Loamy	-0-^{a}	21^{a}	-0-^{a}	-0-^{a}
	Hill	773^{a}	741^{a}	462^{b}	864^{c}
Sumac	Shallow	1976^{a}	1060^{b}	848^{bc}	790^{c}
	Loamy	250^{c}	125^{b}	15^{c}	494^{d}

[1]*Row means followed by different superscripts are significant (P<.05).*

[2]*Mean values represent data from tebuthiuron applied at 1.40 kg/ha active ingredient and for spring burning (March 1987) 4 years after tebuthiuron application.*

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Runoff and Sediment From a Burned Sagebrush Community[1]

J. R. Simanton, G. D. Wingate, and M. A. Weltz[2]

Abstract.—A sagebrush/juniper community was burned at different fire intensities to determine runoff and sediment yields. Runoff was similar between unburned and low intensity burns but almost 4 times greater from high intensity burns. Low intensity burns produced twice the sediment of the unburned. High intensity burning produced 5 times the sediment of the unburned.

Prescribed burning is used on rangelands to: (1) reduce fuel load (2) improve range condition, (3) increase forage, (4) improve wildlife habitat and, (5) increase localized water yield. Fire affects many facets of the natural ecosystem. Watershed response to burning depends on vegetation type (Wright 1974), fire intensity, topography and soils (Rice 1973), season of burning (McMurphy and Anderson 1965), and probably most importantly, climate conditions following the burn. Burning can increase both water and sediment yields on pinyon/juniper dominated rangelands (Roundy et al. 1978) and on chaparral dominated rangelands (Hibbert et al. 1981). Other studies indicate no increase in runoff or sediment from burning mesquite *(Prosopis glandulosa)* or whitebrush *(Aloyisa lycoides)* rangelands and post oak *(Quercus stellata)* savannahs in Texas (Garza and Blackburn 1985, Knight et al. 1983). Researchers have evaluated the hydrologic effect of mechanical and chemical treatments of sagebrush dominated rangelands (Blackburn and Skau 1974, Gifford 1982, Lusby 1979). However, the hydrologic and erosion responses of sagebrush burning have not been evaluated.

This study, as part of the USDA Agricultural Research Service's (ARS) Water Erosion Prediction Project, was to determine runoff and erosion from different aged burns, under two fire intensities, in a sagebrush/juniper vegetation community in northern California.

EXPERIMENTAL DESIGN

Study Site Description

The study site, located in the USDI-Bureau of Land Management's (BLM) Eagle Lake Resource Area in the Susanville District in northeastern California, is typical of the Great Basin sagebrush/juniper vegetation type. Major woody species include big sage *(Artemesia tridentata)*, western juniper *(Junipererus occidentalis)*, bitterbrush *(Purshia tridentata)*, and desert gooseberry *(Ribes velutinum)*. Perennial grasses include Idaho fescue *(Festuca idahoensis)*, western needlegrass *(Stipa occidentalis)*, and squirreltail *(Sitanian hystrix)*. The soil is a Jauriga gravelly sandy loam which is a fine-loamy, mixed, mesic Typic Argixerol with about 35% gravel. The climate is characterized by cold, snowy winters and hot dry summers. Average annual precipitation is 355 mm with 30% occurring during the growing season and 70% as winter snow. The plant growing season begins in early May and continues until mid–July when soil moisture is usually depleted. Livestock grazing is the major land use and the study site had been excluded from grazing one year before and during the study period.

Procedures

Runoff and erosion were measured from plots (10.7 x 3.05 m) under simulated rainfall conditions. Troughs at the lower end of each plot diverted water and sediment into runoff measuring flumes and the hydrograph was recorded by water level recorders. Sedigraphs and sediment yields were determined from periodic water/sediment aliquots taken at the flume's exit.

Rotating-Boom Rainfall Simulator

A trailer mounted rotating boom rainfall simulator (Swanson 1965) was used to apply water to the plots. The simulator has ten 7.6 m booms radiating from a central stem (figs. 1 and 2). The booms support 30 V-Jet 80100 flow-regulated nozzles positioned at various distances from the stem. The nozzles spray continuously downward from an average height of 3 m, move in a circular path over two plots, apply rainfall intensities of

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[2]Simanton and Weltz are Hydrologists, USDA-Agricultural Research Service Aridland Watershed Management Research Unit, 2000 E. Allen Rd., Tucson, AZ 85719; Wingate is Hydrologist, USDI-Bureau of Land Management, Susanville District, 705 Hall St., Susanville, CA 96130.

about 65 or 130 mm/hr and produce rainfall energies of 900-1250 MJ*mm/ha*hr. Rainfall spatial distribution over each plot has a coefficient of variation of less than 10%.

Rainfall Simulation Run Sequence

Rainfall simulations were made on three soil moisture conditions. The dry soil surface run (60 min at 65 mm/hr rainfall rate) was followed 24 hours later by the wet run (30 min at 65 mm/hr rainfall rate) which was then followed 30 min later by the very wet run which had varying rainfall intensity (65 and 130 mm/hr). This sequence provides runoff and sediment data for unsaturated (dry run), field capacity (wet run) and saturated (very wet run) soil moistures.

Treatments

There were 2 natural (undisturbed), 1 clipped, 1 bare, 2 fall 1986 burned (Burn-86), and 2 fall 1987 burned (Burn-87) plots. All the plots were grouped within a 50 by 50 m area with the same soil and vegetation type. The clipped treatment had all vegetation cut to 2 cm height and the clippings removed from the plot. This treatment was used to evaluate plant canopy effects on runoff and erosion and not intended to show grazing effects. The bare treatment had all vegetation clipped to the ground surface and all surface cover (litter, rock and gravel) removed with minimal soil surface disturbance. The Burn-86 plots were burned in the fall of 1986 using a low intensity fire to simulate a prescribed burn followed by overwinter snowpack and high intensity rainfall. The Burn-87 plots were fall burned in 1987 using a high intensity fire just prior to the 1987 rainfall simulations. This burn was to simulate wildfires that are followed by high intensity rainfall. Rainfall simulations were made after plot treatments in the fall of 1987 and again in the spring of 1988 when the clipped and bare plots were retreated. The Burn-86 plots were not reevaluated in 1988.

Vegetation and Plot Characteristics

A 49 pin-point meter was used to measure vegetation composition, canopy cover and height, and ground cover of each plot. Ground cover characteristics included: soil, gravel (5-20 mm), rock (> 20 mm), litter, and basal plant cover. Ten transects across each plot produced 490 readings to describe surface and vegetation canopy cover. Soil moisture content (percent by weight) at 0-5 cm was determined before the dry and wet runs and after the very wet runs.

RESULTS

The plots runoff and sediment yield responses are biased toward exceptional natural rainfall events because during a simulation run each plot was subjected to 1 to 2.4 times the annual rainfall energy for the site (290 MJ*mm/ha*hr). The total rainfall energy applied to each plot during a year's evaluation was nearly 5 times the natural average annual energy.

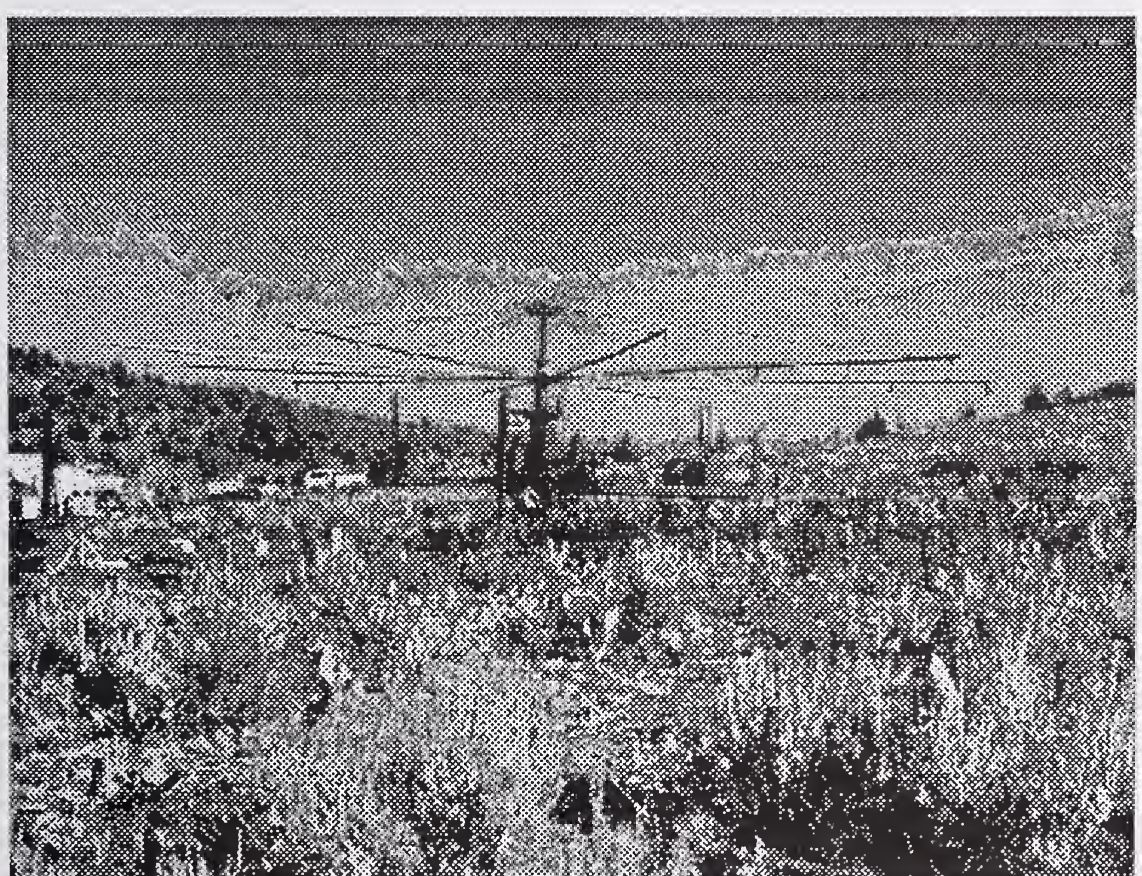

Figure 1.—Rotating boom rainfall simulator at the study site.

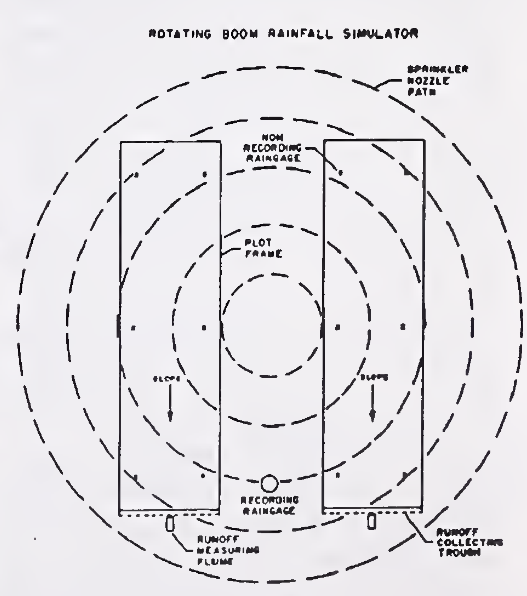

Figure 2.—Schematic of plot layout.

Plot surface and canopy cover characteristics for 1987 and 1988 are presented in table 1. The reason for the changes between the 1987 and 1988 vegetation canopy cover is difficult to explain because of differences in season of measurement. Compared to the natural plots' vegetation composition trend, there was a reduction in the shrub canopy cover component associated with the 1986 and 1987 burns.

Table 2 presents the rainfall, runoff and sediment yield results for all rainfall simulation runs. Runoff and sediment from the natural and clipped plots were similar at all soil moisture conditions. Runoff and sediment yield variability between plots of the same treatment cannot be explained by differences in measured ground or canopy cover. The variablity may be a function of cover distribution on the plot and/or soil variability; factors very difficult to statistically evaluate in natural environments. Because of the small number of plots used in this study, statistical analysis between treatments could not be made.

Sediment yields per mm of runoff from the natural plots were 2 times greater in the fall than in the spring. In contrast, Simanton and Renard (1981) found natural plot sediment concentrations from rainfall simulation studies in shrublands of southeastern Arizona were about 2 times higher in the spring than in the fall.

For each simulation run, the average of each treatment's runoff and sediment yield was divided by the total rainfall of that run. These runoff and sediment yield coefficients were then plotted as a function of soil moisture measured before the beginning of each run (figs. 3-6). Except for the bare treatment, runoff coefficients of the other treatments were similar under low soil moisture conditions (figs. 3 and 5). As soil moisture increased, runoff coefficients of all the treatments increased. The runoff coefficients of the natural, clipped and Burn-86 were similar at all soil moistures. At the very wet soil moisture condition, the runoff coefficient of the Burn-87 treatment was almost 4 times those of the natural, clipped and Burn-86. The bare treatment runoff coefficient increased with soil moisture at a faster rate than the runoff coefficients of the other treatments and at the highest soil moisture condition was over 10 times the runoff coefficient of the natural treatment. This rapid increase of the bare treatment's runoff coefficient is probably a function of increased soil surface crusting and sealing.

The spring 1988 runoff coefficient for the 5-month old Burn-87 treatment was over 6 times the 1988 natural treatment runoff coefficient. Because the natural treatment and 1-yr old Burn-86 treatment had very similar runoff coefficients it appears that the hot burning treatment may either have a long term effect on the runoff response or that a complete growing season is necessary to overcome the fire effects.

Sediment yield coefficients showed trends similar to the runoff coefficients (figs. 4 and 6) with the largest sediment coefficients associated with high soil moistures. The sediment coefficients of the natural and clipped treatments were not different in the fall of 1987. However, in the spring of 1988, one of the natural plots was producing considerably larger amounts of sediment than the other natural and clipped plot. This illustrates the natural spatial variability associated with field sites. The bare treatment had the largest sediment coefficients, especially at high soil moisture (fig. 4). The increase in the bare treatment's sediment coefficient between 1987 and 1988 follows a similar time related trend found for bare plots studied in Nevada and

Table 1.—Vegetation and surface characteristics of runoff plots for fall 1987 and spring 1988.

	Ground surface cover (%)				Canopy cover (%)		
	Soil	Rock/ gravel	Litter	Basal	Grass	Forb	Shrub
Fall 1987							
Natural	14.7	9.0	64.1	12.2	4.9	3.3	21.2
Natural	16.7	16.3	53.5	13.5	4.5	1.6	18.4
Clipped	7.3	9.1	71.8	11.8	0.0	0.0	0.0
Bare	75.7	6.4	7.0	10.9	0.0	0.0	0.0
Burn-86	20.0	21.6	49.4	9.0	0.8	2.0	8.6
Burn-86	19.2	20.4	55.5	4.9	2.9	3.3	8.2
[1]Burn-87	21.2	3.9	51.4	13.5	7.8	1.6	16.3
[1]Burn-87	14.3	18.4	53.1	14.3	7.8	4.1	17.1
[2]Burn-87	40.5	43.3	10.6	0.0	0.0	0.0	0.0
[2]Burn-87	45.0	48.9	11.7	0.0	0.0	0.0	0.0
Spring 1988							
Natural	16.4	14.5	57.7	11.4	13.2	3.6	15.0
Natural	16.4	17.7	55.9	10.0	9.5	1.8	17.3
Clipped	13.2	9.5	70.5	6.8	5.9	0.5	0.0
Bare	66.8	13.6	10.0	9.5	0.5	1.4	0.0
Burn-87	30.0	33.6	33.6	5.5	9.5	4.1	0.0
Burn-87	40.0	27.7	28.2	4.1	7.7	8.6	0.0

[1]*Before burn*
[2]*After burn*

Arizona (Simanton and Renard 1986). Sediment coefficients of both burned treatments were higher than the natural or clipped treatment. At the highest soil moisture condition, the Burn-86 sediment coefficient was about 2 times the natural's. Under similar high soil moistures, the Burn-87 sediment coefficient immediately after burning was 5 times the natural's. Five months after the burning the sediment coefficient of the Burn-87 was 16 times the natural's. In contrast, the sediment coefficient of the bare treatment under the high soil moisture condition was 28 times the natural's in 1987 and almost 220 times greater in 1988 (fig. 4).

CONCLUSIONS

The burning treatments increased runoff and sediment yields from extreme rainfall events on wet soil. Burning will not increase runoff or sediment yields from normal rainfall events when the soil moisture is less than field capacity. Rainfall events occurring on very wet soil, which can occur in early spring, may produce increased yields from burned areas, especially those burned by high intensity fires.

The BLM's policy of prescribed fall burning, using a low intensity fire, appears to be suited for the vegetation community evaluated in this study. The prescribed burning reduces the hazard of wildfires by removing shrub species and accumulated litter and has little effect on surface runoff and sediment yields.

Table 2.—Rainfall, runoff, and sediment from erosion study plots for dry, wet, and very wet soil moisture conditions for fall 1987 and spring 1988.

Treat.	Fall 1987 Rainfall (mm)	Fall 1987 Runoff (mm)	Fall 1987 Sediment (kg/ha)	Spring 1988 Rainfall (mm)	Spring 1988 Runoff (mm)	Spring 1988 Sediment (kg/ha)
			Dry			
Natural	56.1	0.0	0.0	43.7	0.0	0.0
Natural	57.3	0.2	11.6	44.6	0.2	4.7
Clipped	55.1	0.0	0.0	42.2	0.0	0.0
Bare	58.1	4.2	196.0	43.9	7.8	657.2
Burn-86	42.9	0.5	46.4	—	—	—
Burn-86	44.2	0.1	5.3	—	—	—
Burn-87	33.5	0.1	0.0	41.4	0.1	0.6
Burn-87	33.9	1.0	73.2	45.8	0.6	19.3
			Wet			
Natural	24.0	0.4	11.3	23.2	0.1	1.1
Natural	23.4	0.2	17.1	24.6	0.8	19.7
Clipped	26.2	0.4	17.9	22.4	0.1	1.0
Bare	24.9	8.0	[1]500.0	25.2	12.8	3013.3
Burn-86	24.2	0.4	32.3	—	—	—
Burn-86	25.7	0.2	13.0	—	—	—
Burn-87	28.0	1.0	110.4	26.9	2.0	188.2
Burn-87	25.7	2.0	171.1	27.5	2.0	135.5
			Very Wet			
Natural	34.8	1.8	79.2	45.7	0.1	4.8
Natural	26.1	0.7	97.7	27.4	1.3	48.2
Clipped	27.9	1.0	86.1	24.4	0.3	8.7
Bare	28.6	13.7	2369.0	24.9	14.7	4256.1
Burn-86	25.9	1.7	188.8	—	—	—
Burn-86	27.6	0.6	82.5	—	—	—
Burn-87	33.2	3.9	407.5	29.6	6.3	651.2
Burn-87	35.8	5.3	502.6	29.7	5.3	280.9

[1]*Estimated*

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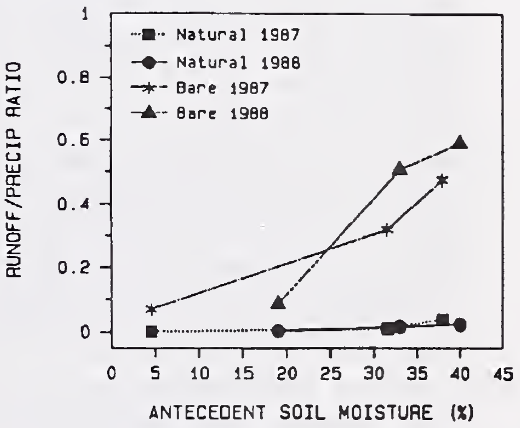

Figure 3.—Antecedent soil moisture vs. runoff/precipitation ratio for the 1987 and 1988 natural and bare plots.

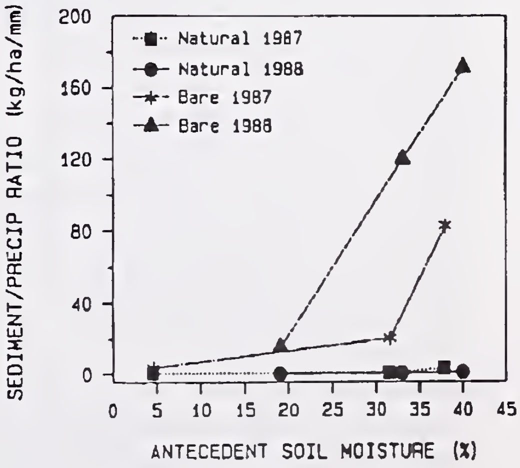

Figure 4.—Antecedent soil moisture vs. sediment/precipitation ratio for the 1987 and 1988 natural and bare plots.

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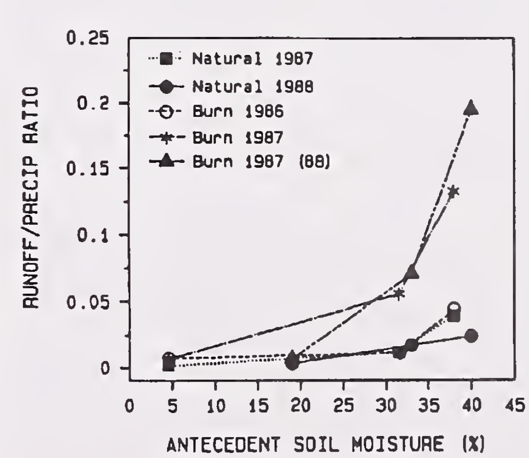

Figure 5.—Antecedent soil moisture vs. runoff/precipitation ratio for the 1987 and 1988 natural and burned plots.

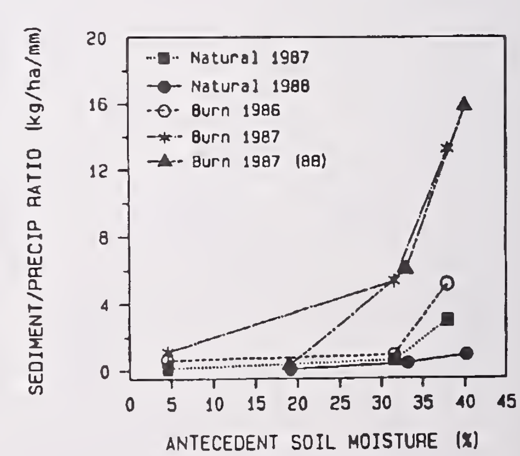

Figure 6.—Antecedent soil moisture vs. sediment/precipitation ratio for the 1987 and 1988 natural and burned plots.

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Seedbed Ecology of Lehmann Lovegrass in Relation to Fire[1]

L. B. Sumrall, B. A. Roundy, J. R. Cox and V. K. Winkel[2]

Abstract.—High seedling emergence of the exotic Lehmann lovegrass *(Eragrostis lehmanniana)* after burning is due mainly to removal of the overstory grass canopy. Canopy removal increases germinability and emergence by changing the light and temperature environment of the seedbed, not by increasing the period of soil water availability.

Lehmann lovegrass (*Eragrostis lehmanniana* Nees.), a drought tolerant, warm season perennial bunchgrass, was introduced to Arizona over 50 years ago (Cable 1971). It now covers over 200,000 ha in southeastern Arizona (Cox and Ruyle 1986). This grass provides important cover for erosion control and forage for livestock on many southwestern rangelands lacking native perennial grass cover. However, Lehmann lovegrass is not evenly utilized by grazing animals due to its stemmy habit (Ruyle et al. 1987) and its invasion into native grasslands may decrease biological diversity (Bock et al. 1986). Fire has been considered as a management tool to address both of these concerns.

Fire may reduce residual lovegrass biomass and improve forage utilization. Although it has been hoped by some land managers that fire might favor native grass establishment at the expense of Lehmann lovegrass, this does not appear to be the case. Lehmann lovegrass seedling emergence increases after burning, especially when mature plants are killed (Cox and Ruyle 1986). Increased seedling emergence is due, in part, to direct effects of the heat of the fire on the seed, possibly by increasing germination through seedcoat scarification (Ruyle et al. 1988). However, burning might also create a favorable environment for seedling emergence by removing the grass overstory canopy. This could result in more favorable light, soil water, and temperature conditions for Lehmann lovegrass seed germination and seedling establishment. An understanding of fire effects on the seedbed environment in relation to establishment requirements is necessary to determine how fire or other management practices may be used to either favor or disfavor persistence of Lehmann lovegrass. The purpose of this ongoing study is to determine the effects of fire on the seedbed environment that influence seedling emergence of Lehmann lovegrass.

Study Site and Methods

The study is being conducted on the Santa Rita Experimental Range 60 km south of Tucson, Arizona. The study site is at 1200 m elevation and supports a nearly pure stand of Lehmann lovegrass. Annual precipitation averages 398 mm with 60% falling between June and September. The soil is a Comoro fine sandy loam, Typic Torrifluvent.

Treatments were structured to compare effects of eliminating the canopy and eliminating competition for water by mature plants with the direct effect of burning on seedling emergence. The area was divided into 4 blocks, each containing 8 plots 7.5 by 15 m in area. Treatments were assigned to plots in a randomized complete block pattern (table 1). For the control treatment, mature plants and grass canopy were left intact. The burn treatment consisted of burning in November and spraying with glyphosate to kill surviving mature plants in April. This resulted in an initial seedbed heat treatment and elimination of the grass canopy. The clip treatment consisted of mowing mature plants to a 5-cm stubble height in November and spraying with glyphosate to kill mature plants

Table 1.—Treatments applied to a stand of Lehmann lovegrass at the Santa Rita Experimental Range.

Treatment	Expected effect on seedbed: Light	Water	Initial heat treatment
Control (control)	-	-	-
Burn Nov., spray (with glyphosate) (burn)	+	+	+
Clip and spray (clip)	+	+	-
Spray only (dead standing)	-	+	-

[1]Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).

[2]Sumrall and Winkel are graduate research assistants and Roundy is assistant professor, School of Renewable Natural Resources, University of Arizona, Tucson, Arizona. Cox is range scientist, USDA/ARS, Tucson, Arizona.

in April. This resulted in an elimination of the grass canopy but without an initial heat treatment. The dead standing treatment consisted of spraying live mature plants in April to kill them and eliminate their use of soil water while leaving the dead canopy intact.

To determine pre-emergence germinability of seeds in the seedbank, 8 bioassay samples, 5 by 6 cm in area and 1-cm deep, were collected from each plot prior to summer rains in July. All samples were watered in the greenhouse and number of emergent seedlings recorded.

Seedling emergence was monitored regularly through the growing season starting after consistent summer rainfall occurred in July. Seedling density was quantified in 20 0.25–m², permanently-marked quadrats per plot. Soil temperatures, incident solar and net radiation, and soil water potential were measured with thermocouples or thermisters, pyranometers and net radiometers and gypsum blocks, respectively, and recorded using electronic microloggers. Measurements were recorded for selected periods in fall, winter, and spring, and were continuously recorded during the summer rainy season.

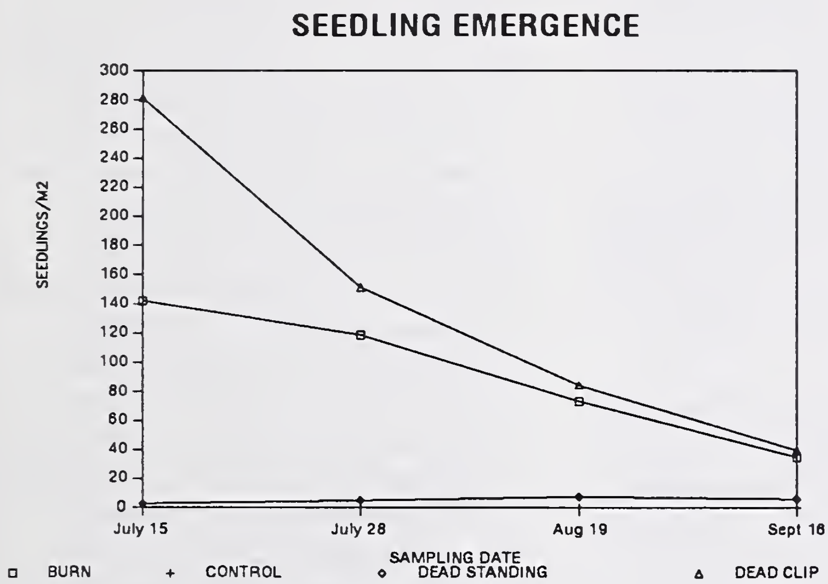

Figure 1.—Seedling densities of Lehmann lovegrass with overstory grass canopy intact and alive (control), intact but dead (dead standing) or removed by burning (burn) or clipping (dead clip).

Results and Discussion

Bioassay samples taken in July indicate that there were numerous germinable Lehmann lovegrass seeds in the seedbed prior to the summer rains. Bioassay samples produced 2,385; 1,198; 573 and 1,010 seedlings/m² on clipped, burned, control and dead standing canopy treatments, respectively. Plots where the grass canopy was removed by clipping had significantly ($P \leq 0.05$) more germinable seeds than other plots. Field plots where the canopy was removed, either by clipping or burning, had significantly ($P \leq 0.05$) greater seedling emergence than those with an intact canopy (fig. 1). The percentage of germinable seeds (as indicated by bioassay) which emerged (as indicated by maximum field seedling density on 14 July) was 11.8, 11.9, 0.02 and 0.26 for clipped, burned, control and dead standing canopy plots, respectively. Removing the canopy apparently not only increased germinability of seeds in the seedbank, but also increased the number of germinable seeds that emerged as seedlings.

Removal of the canopy may change the soil water, temperature and light conditions of the seedbed. Soil water potential was high on all plots during the period of maximum seedling emergence. During a subsequent drying period, water potentials decreased more rapidly on burned

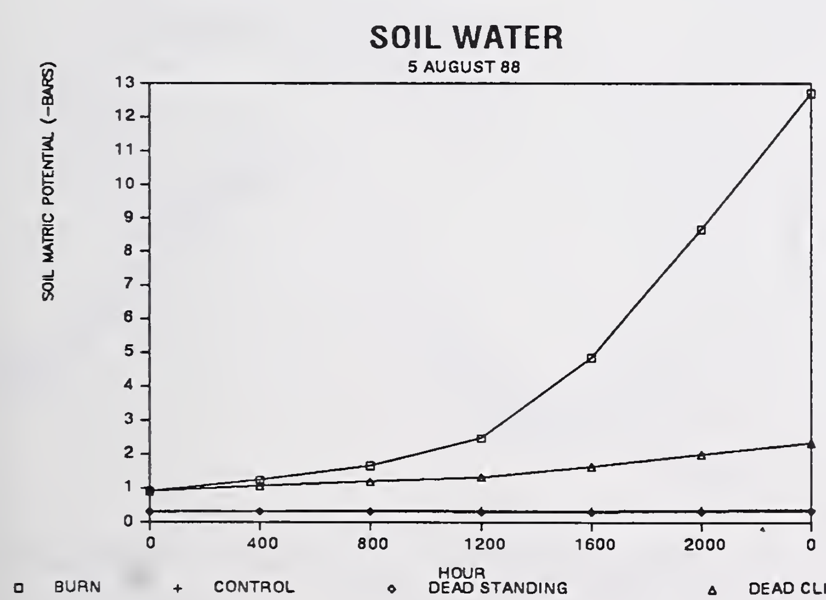

Figure 2.—Soil matric potential at 1-3 cm during a drying period after summer rains in a Lehmann lovegrass stand with the grass canopy intact and alive (control), intact but dead (dead standing) or removed by burning (burn) or clipping (dead clip).

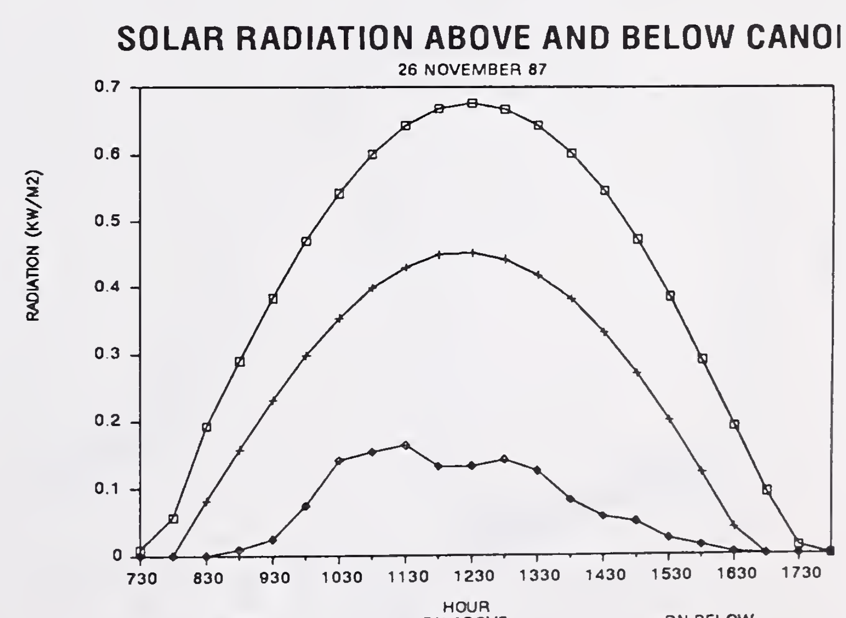

Figure 3.—Incident (RT) and net (RN) radiation above and below a Lehmann lovegrass canopy.

and clipped plots than plots with either a live (control) or dead canopy (dead standing) (fig. 2). Evidently, soil water was lost more to evaporation than transpiration, and soil water availability did not account for differences in seedling emergence.

The grass canopy affects both the quantity and quality of light reaching the seedbed. In November, when the canopy was senesced, net radiation at the top of the canopy was about 67% of the incident solar radiation, but net radiation under the canopy was less than 24% of the incident solar radiation (fig. 3). Thus, at least 33% of the incident radiation was reflected by the canopy while 43% of incident radiation went to heating the canopy and did not reach the seedbed. The lower input of energy to the seedbed resulted in lower soil temperatures and a smaller diurnal difference in soil temperatures where the canopy was intact than where it was removed by clipping or burning (figs. 4 and 5). The warmer seedbed and greater diurnal temperature differences on burned and clipped plots may have increased germinability of seeds in the seedbank, possibly by breaking down the seedcoat. Germination tests of newly harvested seed from this site indicated that only 5% were readily germinable but that an additional 45% would germinate when the seedcoat was mechanically scarified. A greater range in diurnal temperatures may increase germination of warm season grasses during optimum soil water conditions. Saltgrass *(Distichlis spicata)* requires a 20°C differential in diurnal temperatures for maximum germination (Cluff and Roundy 1988).

The grass canopy may also modify the quality of light reaching the seedbed during summer rains when temperature and soil water conditions are optimum for germination. Green leaves absorb strongly the red and blue wavelengths so that below-canopy light has a low red/far-red ratio (Bewley and Black 1982). Predominant red light may induce dormancy by changing the phytochrome ratio in the seed and may inhibit germination by reducing cell elongation in the radicle (Bewley and Black 1982).

The effects of quality of light and alternating temperatures on germination of Lehmann lovegrass are subjects for future research. The present study indicates that although heat from fire may directly increase germinability of Lehmann lovegrass, the removal of the canopy by fire is most responsible for increased seedling emergence after burning. These results also suggest that seedling establishment could be much greater in heavily-grazed than ungrazed stands of Lehmann lovegrass and are of use in determining how to manage for persistence or replacement of this exotic species.

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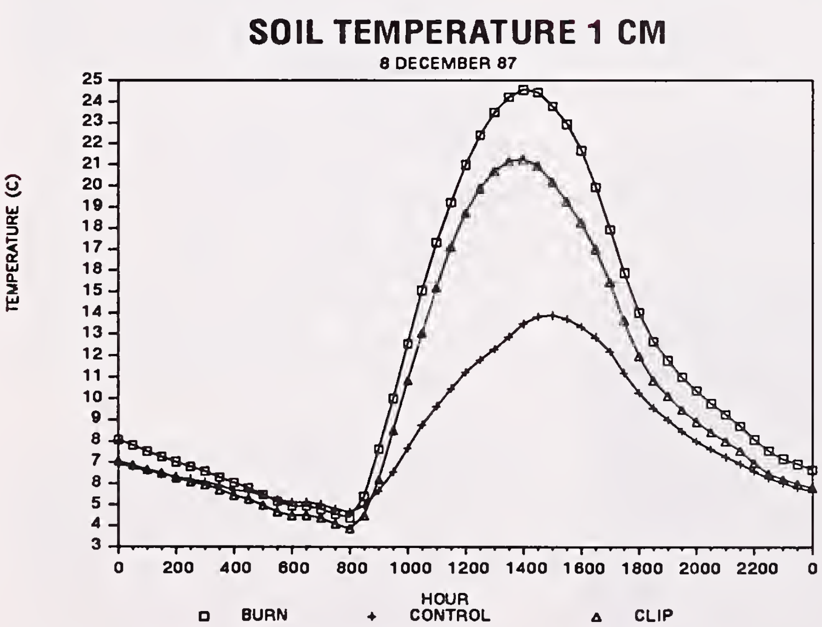
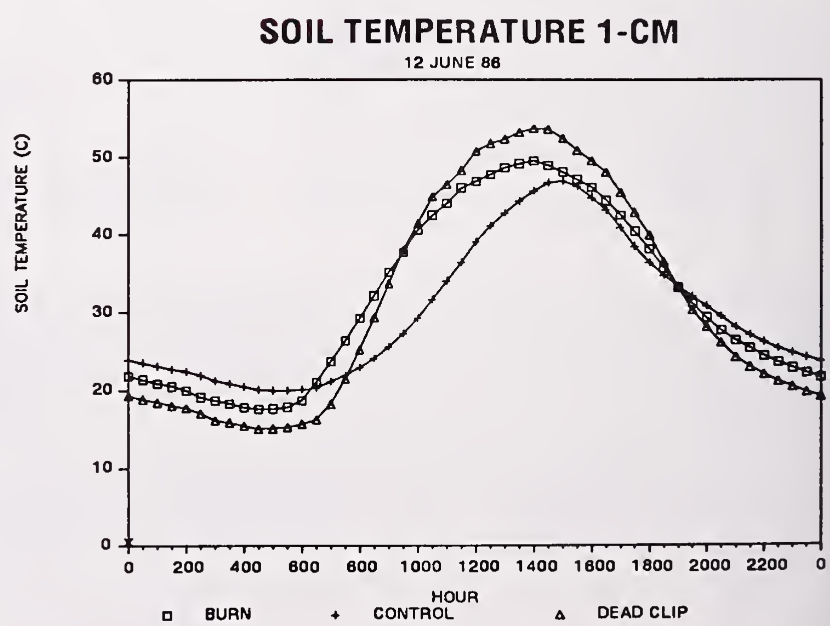


Figure 4.—Winter soil temperatures at 1-cm in a Lehmann lovegrass stand with the grass canopy intact (control) or removed by burning (burn) or clipping (clip).



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Fire Management in the Taos-Blue Lake Wilderness Area[1]

Thomas V. Skinner[2]

Abstract.—Congress established the Taos-Blue Lake Wilderness Area to preserve an area sacred to the Taos Indians in a natural state. Because lightning-caused wildfires are natural and because a natural wilderness is sacred to the Taos Indians, the BIA and Tribal officials have jointly developed a wildfire management plan that permits lightning fires to run their course.

Introduction

The Bureau of Indian Affairs (BIA), in conjunction with Taos Tribal officials, has developed a wildfire management plan to permit wildfires of natural origin to run their course within portions of the Taos-Blue Lake Wilderness Area. Congress established the Taos-Blue Lake wilderness (fig. 1) in 1970 to safeguard the interests and welfare of the Taos Indians because the Taos Indians have depended upon this area "...for water supply, forage for domestic livestock, and as the scene of certain religious ceremonials (84 Stat. 1437)." This paper describes the process undertaken to develop this wildfire management plan and the objectives for wildfire management in the Taos-Blue Lake Wilderness.

Background

The Taos Indians have occupied the same region of northern New Mexico for centuries, building a series of pueblos and using the adjacent forest land for its various resources. Blue Lake, located at the headwaters of the Rio Pueblo de Taos, is sacred to the Taos Indians and is the site of certain religious ceremonials (Collier 1949). The land surrounding Blue Lake had remained in the Public Domain until President Theodore Roosevelt withdrew land to establish the Taos Forest Reserve (34 Stat. 3262). Because the Taos Indians protested this withdrawal of their sacred land, Congress established a special use area that included Blue Lake to protect the Taos Indians' religious interests (48 Stat. 108). However, the Taos Indians decided that this special arrangement insufficiently protected their sacred land and they initiated a process to regain ownership of their sacred land. In 1970, Congress enacted legislation transferring 48,000 acres of land to the BIA specifying that the land be managed according to the Wilderness Act of 1964 (84 Stat. 1437). The wildfire management plan described in this paper is the first complete attempt to actively manage the Taos-Blue Lake Wilderness Area in compliance with the Wilderness Act and in conjunction with Taos Tribal officials.

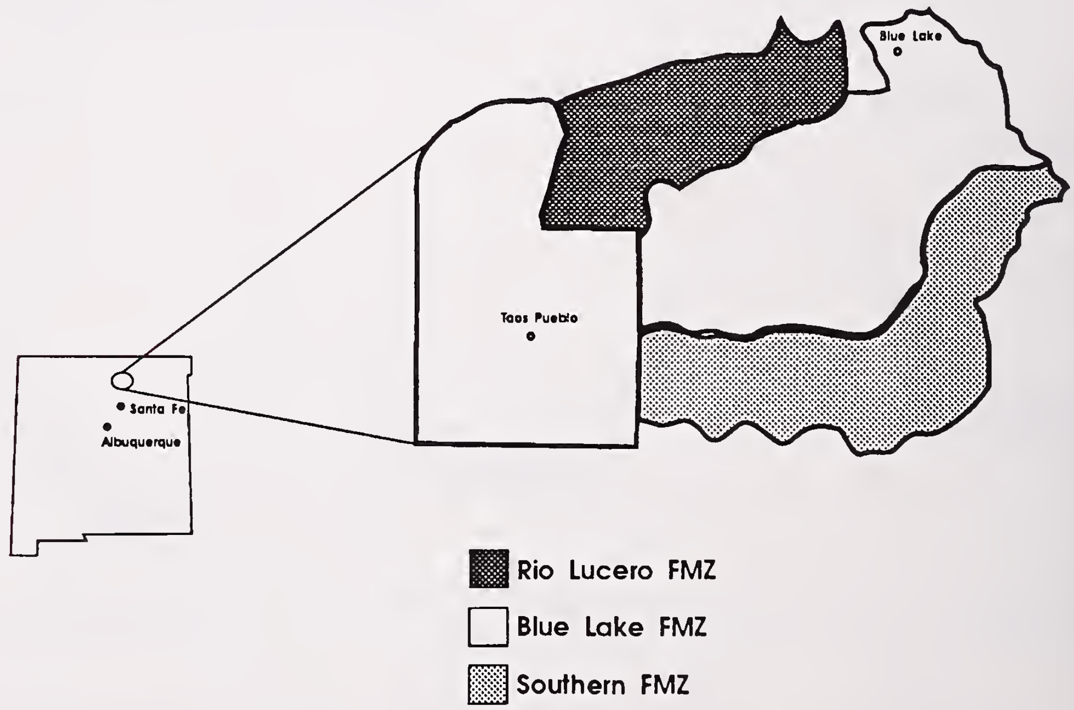

Figure 1.—Forested portion of the Taos Indian Reservation showing the Taos-Blue Lake Fire Management Unit and the three fire management zones: Rio Lucero, Blue Lake, and South-

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Temporary Forester, USDA Forest Service, Southern Forest Experiment Station, Starkville, MS.*

Management Plan Development Process

The Taos-Blue Lake Fire Management Plan (Skinner 1988) was developed in conjunction with tribal officials over a span of two years. Personnel from the BIA along with Tribal officials resolved to develop a plan to manage wildfires in the forested portions of the Taos Reservation. The Taos Reservation consists of six tracts of land; about half of the land is forested. Both BIA and Tribal officials decided that the Wilderness Area (land transferred back to the tribe in 1970) and the land designated as Tract C should be considered together as the Taos-Blue Lake Fire Management Unit (fig. 1). The Unit was subdivided into three fire management zones: Rio Lucero, Blue Lake, and Southern (fig. 1).

I collected and analyzed a variety of data on: fire history, fuel loading, current forest structure, and others. The following conclusions were made: fires had been occurring frequently (average fire return intervals range from 5 to 20 years) during the past several centuries but have become infrequent during the 1900's; fuel conditions fit the standard fuel models; modeled fire behavior will permit fires to burn under prescriptions that keep fire behavior below a four foot flame length threshold (Deeming et al. 1977).

The BIA and Taos Tribal officials used these conclusions to develop two management strategies for the three fire management zones. The Blue Lake Zone (fig. 1) would retain a total suppression posture to protect the sensitive religious sites within this zone. The remaining zones would be subjected to a modified suppression policy for natural wildfires. This modified suppression would be determined by an annual process of ratifying the wildfire management plan with Tribal officials and by a case-by-case, daily analysis of wildfires that have ignited within the modified suppression zones.

Results

I prepared a two-part wildfire management plan: part one, an operational plan, and part two, background data. The operational plan will be subjected to annual review while the background data, which consists of seven documents (covering fire management objectives, general history, fire history, fire ecology-fire effects, geographic information, burning prescriptions, and a fire situation analysis) will likely remain unchanged.

The operational plan not only addresses the issues in wildfire management but also refers to agency policy statements instead of reproducing them. Simply put, the operational plan states the policies that permit natural wildfire management, designates where natural wildfires will be permitted, indicates how wildfires will be responded to, and how wildfires will be managed. I intended the operational plan to be evaluated annually and submitted for approval to the Taos Pueblo War Chief (the Tribal entity who governs land use in the Wilderness Area) prior to the beginning of the fire season; it terminates annually at the end of the calendar year.

The background data sections are independent documents that will permit interested readers to concentrate on whatever subject area they are interested in while ignoring other sections. I anticipated that background data would remain static, needing only periodic amendments.

The BIA and Tribal officials developed the following fire management goals and objectives.

Goals: Prevent fires from impacting ceremonial sites and paths used to reach these sites, and minimize participation of non-tribal members in fire suppression in the wilderness area during ceremonial periods.

Objective: Protect religious and ceremonial sites, identified by the Taos Pueblo War Chief, and coordinate with the Taos Pueblo War Chief during ceremonial periods to manage fire solely with Tribal members.

Goal: Manage wilderness for natural values by allowing fire to play a natural role in the wilderness.

Objective: Permit fires to burn within the modified suppression zones under pre-specified conditions and under close monitoring to perpetuate natural vegetation patterns and mosaics.

Goal: Use fire to accomplish desired management objectives.

Objective: Establish management objectives for the wilderness area (e.g. create and maintain habitat for desired wildlife species, reduce hazardous fuel accumulations) on an annual basis.

Goal: Protect human life from unwanted fire.

Objective: Coordinate with the Taos Pueblo War Chief any agreed upon closure within the wilderness area to protect wilderness users.

Goal: Protect property from unwanted fire.

Objective: Prevent the spread of fires from Taos Pueblo reservation Land onto private or USFS land unless a prior agreement has been made to permit such a crossing.

Goal: Avoid unacceptable fire effects.

Objectives: Suppress fires within the modified suppression zone that cause unacceptable resource impact. Maintain acceptable air quality. Minimize fire caused sedimentation by aggressively suppressing fires within 100 yards of Rio Pueblo de Taos and La Junta Creek.

Goal: Minimize impacts of suppression activities.

Objectives: Use only those tactics required to suppress fires. Use fire retardants only when necessary. Avoid using mechanized equipment to suppress wildfire. Prevent erosion from constructed firelines.

The plan was submitted to the

Taos War Chief for approval and went into effect on 4 April 1988. The plan will be subjected to an annual approval process and remains operational until the end of each calendar year.

Discussion

The Taos-Blue Lake Wilderness Area is the only Congressionally identified wilderness area within the BIA (Tandy 1985). Because a natural state is important for the religious practices of the Taos Indians and because the transferring Act mandates wilderness management within the Wilderness Area and because competent wilderness management should include consideration of naturally-caused fires, the BIA developed this fire management plan.

I structured the plan in two parts because I felt that although the background data are important, they should not encumber the annual evaluation and implementation of the operational plan. However, the background data can be appended as data becomes available. For example, fire management objectives should be evaluated periodically as should the fire situation analysis procedure. However, fire history data, fire ecology information, and geographic information are unlikely to change.

Acknowledgments

The Taos-Blue Lake Fire Management Plan was prepared with special project funding from the Boise Interagency Fire Center, Northern Pueblos Agency, and Albuquerque Area Office, all for the Bureau of Indian Affairs. The plan was prepared while the author was a temporary Forester at Northern Pueblos Agency. Preparation of this paper was done solely at the author's expense. Presentation of this paper was supported by both the BIA and the USFS. This plan has had the support of the Taos Pueblo War Chiefs and members of their staff.

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Forest Fires in Mexico: Causes and Strategies[1]

Luis Antonio Bojórquez-Tapia[2]

Abstract.—Most forest wildfires in Mexico are related to human activities. In general, the causes include a combination of economic, social, and cultural factors. A new strategy has been implemented since 1984, which includes the use of modern technology, improvement of socio-economic aspects, and use of alternative natural resource management schemes.

Forests and woodlands in México cover 38.9 million ha, approximately 20% of the territory (SARH 1988). They are diverse communities from the taxonomic and ecological viewpoints (Rzedowski 1978). However, despite the abundance of natural resources, forestry has been relegated to a secondary role in the Mexican economy by a complex combination of cultural, economic, and social factors. As an aftermath, deforestation is rampant; about 5.3 million ha have been lost since 1978 (SARH 1980, 1988). Misuse of fire as a management tool has been an important element contributing to forest devastation. Therefore, it can be asserted that one of the effects of those factors is the high incidence of man made forest fires in México.

The objective of this paper is to summarize the causes of wildfires and the strategies proposed and implemented to prevent and combat forest fires in México.

It is estimated that approximately 90% of all wildfires are associated to human activities (SARH 1985). Although the main causes of wildfires are regional (table 1), wildfires are induced by one or a combination of several of the following socioeconomic and cultural reasons (Rzedowski 1978):

1. Inadequate management schemes makes forestry unprofitable for poor rural inhabitants. This motivates rural inhabitants to look for relatively better-paying occupations. But, scarcity of alternative employment force them to agriculture or cattle raising with meager yields. Fire is used for clearing the land and forage improvement. Ordinarily, fires are lighted every year in most inhabited forest lands, which can generate acute ecologic changes when combined with overgrazing (Benítez-Badillo 1988). However, forest residents are compelled to move to untouched forested areas because improper agriculture, overgrazing, and wildfires produce soil erosion and general environmental degradation in short time.

2. Nomadic agriculture is traditional in the tropical regions. It consists of clearing, burning, planting, and growing crops in an area for several years. For the Mayans, nomadic agriculture was a multiple-use method that allowed tropical forests to recover after a period of crop production (Toledo et al. 1976). Unfortunately, this agricultural method is obsolete since land recovery is not permitted nowadays, partly because of the excess of rural population in relation to the available land.

3. Unchecked agricultural crop fires that burn free into adjacent forests.

Table 1.—Regional causes of wildfires in México (SARH 1985).

Region & State	Cause by Region
Northwest	
Sonora	grassland burns
Baja California	campfires
Baja California Sur	lightning
North-central	
Chihuahua	campfires
Durango	sawmill operations, logging operations
Northeast	
Coahuila	crop fires
Nuevo León	grassland burns
Tamaulipas	lightning
San Luis Potosí	
Central	
Jalisco	grassland burns
Edo. de México	crop fires
Distrito Federal	arson fires
Morelos	campfires
Tlaxcala	agricultural clearings
Puebla	
Hidalgo	
Southeast	
Veracruz	nomadic agriculture
Chiapas	clearings for cattle raising
Oaxaca	
Tabasco	
Quintana Roo	
Campeche	

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Centro de Ecología, UNAM, Apartado Postal 70-275, México, D.F. 04510, México*

4. Land ownership and use are anarchical in some regions; when conflicts arise, arsonists burn for revenge or spite. In relation to the design of management programs, this factor also complicates planning and implementing forestry programs.

5. Public involvement in fire prevention and combat is insignificant because the role of forests in providing goods and services, besides timber and firewood, has not been recognized by the general population. Consequently visitors' negligence provokes wildfires in areas where camping, hunting, fishing, and picnicking are popular.

6. Increasing demands for natural resources, living places, or recreational areas by growing rural and urban populations.

7. Limited resources are available to activities for wildfire presuppression and combat. The only governmental office assigned to these activities is the Secretariat of Agriculture and Hydraulic Resources, which is poorly coordinated with other governmental offices.

Strategy

Considering that human activities are the main cause of wildfires, a new strategy has been implemented for presuppression and combat since 1984. This strategy consist of the following programs:

1. Governmental agencies coordination.—To optimize available resources, it has been necessary to coordinate with different governmental offices to locate, prevent, and combat wildfires. Formal agreements have been established between SARH and local and federal authorities; outstanding accords have been subscribed with the Civil Aeronautic Bureau, to help in locating fires, and the Secretariat of Defense, for logistical and material support, and personnel.

2. Civic co-partnership.—The objective of this program is to increase public understanding and cooperation. Establishment of close contacts between SARH and local civic associations are encouraged. The latter include, for example, ecologists, boy and girl scouts, amateur radio clubs, and agricultural, forestry and ranching societies. A series of meetings are implemented to educate the public about the effects of wildfires, increase public interest with local information, develop proper public attitude and opinion concerning wildfires, and demonstrate correct procedures for preventing and dealing with fire.

3. Private enterprises co-partnership.—This program looks for the collaboration of forestry private enterprises in fire prevention and suppression; specially by providing funds, personnel, and supplies.

4. Training and technological development.—The National Center of Training and Technological Development was founded to achieve proper training of firemen and improvement of fire fighting tools and equipment.

5. Advertising.—The objectives of this program are similar, though more general, to those of the program for civic co-partnership. Mass media campaigns, including radio, television, newspapers and other publications, has been implemented to reach the general public. Each of these media devotes a quota of its time or space to community service free of charge.

These programs are coordinated through an Operating System, which was devised to integrate prevention and combat policies. The operating system is divided into presuppression and combat. Presuppression combines the six programs with other prevention procedures such as fire management, installation of lookout towers, establishment of firebreaks, and introduction of appropriate natural resources exploitation methods. Combat consists of a follow through procedure for detection, planning attack (based upon reports and meteorological data), control and suppression tactics, and assessment of damages.

Results

Both the total annual number of wildfires and the total annual area affected per year have increased since 1961 (figs. 1a-b). At least partially, these results can be explained by the implementation of better fire detection procedures (Martínez 1988). Although the number of fires and area affected are correlated to each other, the average area affected per fire has decreased from 1983 to 1987 (fig. 1c), which is significant because during the period 1986-87 the number of fires and the area affected recorded were the greatest (figs. 1a-b). This indicates that the Operating System has worked satisfactorily, at least in wildfire combat. Notwith-

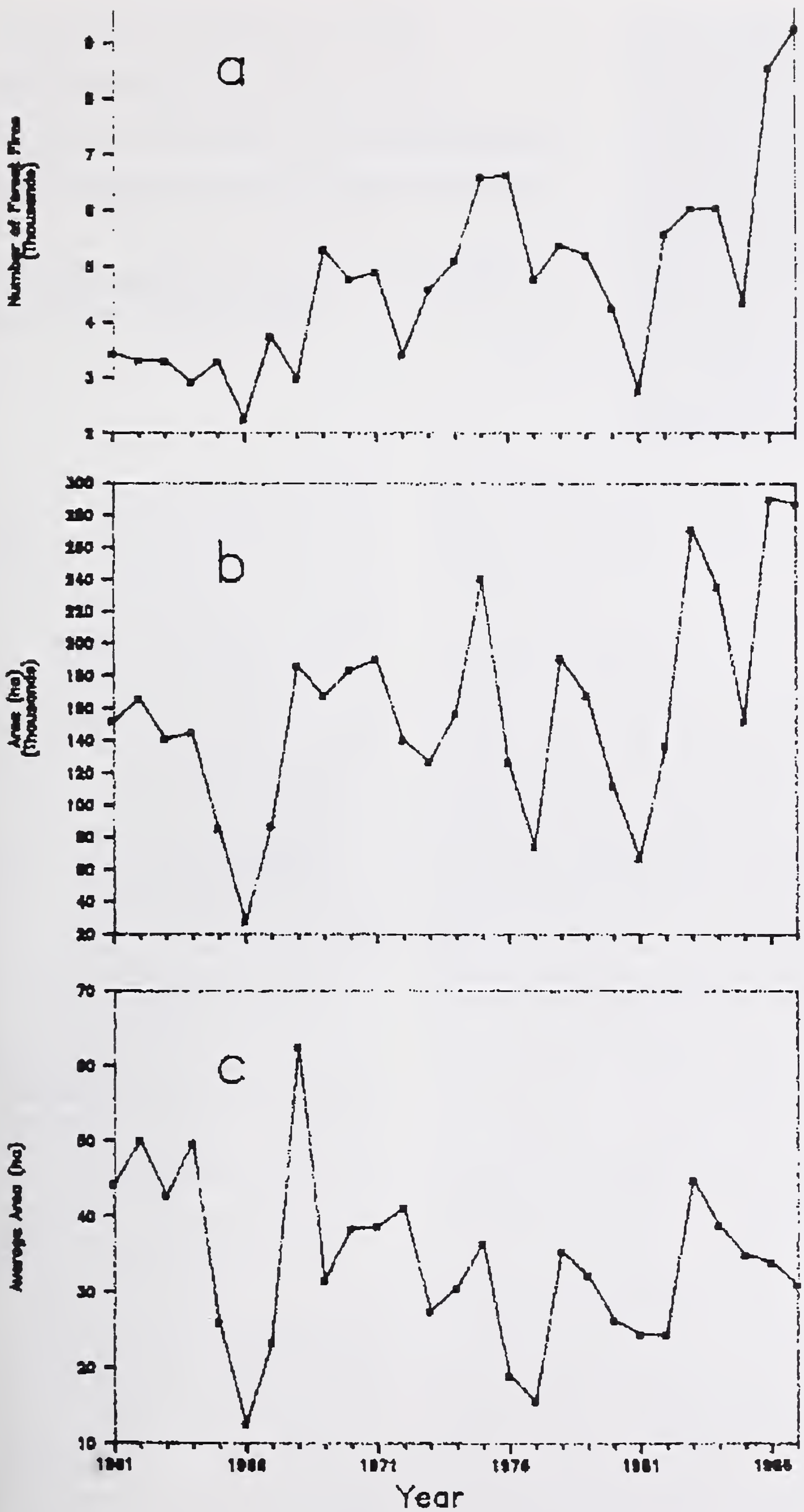

Figure 1.—Wildfires in México: (a) total number of wildfires per year; (b) total affected area per year; (c) average area affected per fire per year.

standing, general population's quality of life has to improve before the goals of wildfire presuppression programs are achieved.

Limitations and Future

Although the results are promising so far, serious limitations exist for fulfilling the goals of the Operating System. First, the economic crisis in México has limited the funds available for fire presuppression and combat. Second, meterological data are not sufficient for fire combat planning. Finally, compilation of reports is time consuming and inaccurate.

Consequently, those three areas have to be resolved in the future. Better meterological data is needed for fire fighting and limiting access to forests during severe dry periods. A computerized data base will be indispensable for prediction and assessment of damages.

Acknowledgments

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Impact of Fire on the Microbial Processes in Pinyon-Juniper Woodlands: Management Implications[1]

Carole Coe Klopatek, Leonard F. DeBano, and Jeffrey M. Klopatek[2]

Abstract.—Plants grown in soils burned when dry had a lower VAM colonization than soils burned when wet. Juniper soils demonstrated the greatest reduction (>95%). Plants grown in interspace soils burned when wet were least affected. Unburned control soils from interspace and juniper soils contained significantly larger nitrifying bacteria populations than pinyon soils. Nitrifying bacteria were less affected by burning when wet than dry, again with juniper soils showing the greatest reduction (over 80%). Positive correlations were found between the decrease of both VAM and nitrifying bacteria, and resulting soil temperatures.Temperature effects and associated reductions in VAM and nitrifying bacteria were related to amount of litter burned in each microcosm and the moisture content of the soils.

Mycorrhizae are mutualistic (symbiotic) relationships formed between fungi and plant roots of a host plant (Harley and Smith 1983). The fungi serve as nutrients and as water-absorbing organs for the plant (Menge et al. 1978, Powell and Bagyaraj 1984, Safir 1987), whereas the plant provides photosynthate (carbon source) for the fungi (Mosse 1973). A second significant contribution the fungi play is protection against invading parasitic organisms (Davis and Menge 1981, Powell and Bagyaraj 1984, Schonbeck and Dehne 1977). There are two main divisions of mycorrhizae: ectomycorrhizae which are associated with coniferous forest trees such as pine, spruce, and fir; and endomycorrhizae which are formed by the majority of members of all land plants, (e.g., ferns, grasses, cacti, shrubs, and trees). Although the study of mycorrhizae is fairly recent, the relationship between plant and fungi is not. In fact, one of the first land plants, Rhynia (Chaloner 1970), was found to be colonized by mycorrhizae (Kidston and Lang 1921, Nicolson 1975).

It has been well documented that plants are strongly dependent on mycorrhizal fungi. Without this symbiosis many plants fail to grow past the germination stage or show a decreased growth rate (Harley 1969, Harley and Smith 1983, Mosse 1973, Powell and Bagyaraj 1984). Many studies have shown that this relationship is fragile and can be easily disturbed (Bethlenfalvy and Dakessian 1984, Daft and Nicolson 1974, Janos 1980, Klopatek et al. 1988, Reeves et al. 1979, Warner 1983, Williams and Allen 1984). These studies show that as the severity of disturbance increases, i.e., from livestock grazing (Bethlenfalvy and Dakessian 1984, Reece and Bonham 1978) to surface mining (Allen and Allen 1980, Gould and Liberta 1981, Zac and Parkinson 1982), there is a corresponding decrease in the frequency of VAM propagules and/or colonization.

The pinyon-juniper woodlands are the third largest vegetation type in the United States (Klopatek et al. 1979). In Arizona, approximately 5.75 million hectares (Arnold et al. 1964) are covered by this woodland. This ecosystem is unique in that it is characterized by having codominants of different mycorrhizal status. Pinyon is ectomycorrhizal and juniper is colonized by endomycorrhizae, more specifically vesicular-arbuscular mycorrhizae (VAM). This woodland is currently undergoing a number of different perturbations, including fuelwood harvesting, intensive cattle grazing, and stripmining for coal. Additionally, both wild and prescribed fires are reoccurring phenomena in pinyon-juniper (USDA Forest Service 1981). The majority of these fires are prescribed to eliminate the trees and allow for greater grass production for grazing.[3] The trees are thought to out-compete grasses for water and nutrients. Although this woodland has both endomycorrhizal and ectomycorrhizal components, we chose to investigate only the vesicular-arbuscular endomycorrhizal component because the majority of plants present in mature pinyon-juniper communities are colonized with VAM and therefore they are probably important for recovery after disturbance.

There have been a number of studies on the nutrient distributions in pinyon-juniper woodlands (Charley and West 1975, Everett et al. 1986, Klopatek 1987). Nutrients have been shown to be dispersed in what are coined "nutrient islands" having a canopy-covered, nutrient-rich area and a nutrient-poor interspace region (Barth and Klemmedson 1978, Char-

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Microbial Ecologist USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287-1304 and Department of Microbiology, Arizona State University, Tempe, AZ 85287-1601; Supervisory Soil Scientist, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287-1304; and Associate Professor, Department of Botany, Arizona State University, Tempe, AZ 85287-1601.*

[3]*USDA Forest Service, Tusayan, AZ (personal communication).*

ley and West 1975, Klopatek 1987). The cycling of an essential nutrient, nitrogen, is controlled by microorganisms (Alexander 1977). We chose to investigate one important part of this cycle, nitrification. It has been shown that after a major disturbance, such as fire, there is a significant increase in nitrification (Vitousek and Matson 1985). Production of NO_3 presents a potential loss of N from the ecosystem because NO_3 is easily leached (Brady 1974). Given the potential increased production of NO_3 produced after fire, we wanted to examine what effects fire has on the organisms responsible for this process.

Thus, the objectives of this study were twofold: to determine whether VAM propagules are reduced following fire and whether fire affects the propagules' ability to colonize host plants and, secondly, to determine the immediate response and subsequential recovery of nitrifiers following fire.

MATERIALS AND METHODS

Site Description

Soils for this study were collected from a pinyon-juniper community, referred to as the Dillman site, 30 km south-southeast of Grand Canyon, Arizona, in the Kaibab National Forest. *Pinus edulis* Engelm. is the pinyon pine species in this area and the codominant juniper is *Juniperus osteosperma* (Torr.) Little. The dominant interspace and understory species is blue grama grass [*Bouteloua gracilis* (H.B.K.) Lag.]. Site elevation is approximately 2030 m. Soils are classified as Lithic Ustochrepts, which were derived from Kaibab Limestone. Precipitation averages 350 mm/yr, occurring during summer as convectional thunderstorms and during the winter from frontal systems. Annual mean monthly temperatures range from -1.4° to 21.1°C, with a mean temperature of 9.2°C. Further site descriptions can be found in Klopatek (1987).

Experimental Design

To account for spatial variability, materials were collected from beneath pinyon and juniper trees and interspaces on three sites in the study area. On each site, five subsamples of soil (to a depth of 10 cm), duff, and litter were collected by shovel and composited separately from beneath pinyon and juniper trees. Composite samples of only litter and soil were collected from interspaces because no duff was present. In total, 24 separate composite samples consisting of 9 soil, 9 litter, and 6 duff samples were collected for study. Soil, duff, and litter were used to reconstruct laboratory microcosms for pinyon trees, juniper trees, and interspaces in 35-cm-high clay irrigation pipes having an inside diameter of 20 cm. The bottoms of the pipes were sealed and a 10-cm layer of sterilized, acid-washed sand (containing < 1 mg kg^{-1} inorganic N and 1 mg kg^{-1} PO_4) was placed in the bottom of each pipe. The sand was covered with filter paper to separate it from the soil above. Nine microcosms were constructed for each of the three cover types (9 pinyon, 9 juniper, 9 interspaces) for a total of 27 individual microcosms. In one series (nine microcosms), three replicates from each cover type, were wetted with sterile distilled water to reach 50% field capacity, prior to burning. In the second series, nine microcosms were burned having soil moisture content similar to that of field-dry soils (approximately 10% on a weight basis). The third series of microcosms, also at field-dried moisture conditions, were not burned and represented the controls.

Temperature probes were inserted at seven different depths (litter, duff, soil surface and soil depths of 1, 2, 5, and 10 cm) to record fire temperatures (complete temperature profiles are reported in DeBano and Klopatek 1988). A fire was simulated on top of each microcosm by using six 500-watt infrared heat lamps to produce instantaneous ignition of the litter. Each treated microcosm was heated with the lamps for a period of 15 min. All litter was consumed by fire within this period, and those microcosms containing duff continued to smolder for several hours depending on the moisture content of the soils. Samples were collected from the 2-5 cm soil depth of the microcosms 24 h after ignition. All microcosms were brought up to 50% field moisture capacity after the first sampling period (to maintain uniform environmental conditions).

Mineral soil samples taken previous to burning were analyzed for organic carbon (Walkley-Black), total Kjeldahl N and phosphorus (Olsen and Sommers 1982), and pH (1:1 soil:water) and soil texture by hydrometer (Day 1965) (table 1). No statistical differences due to burning were noted in these soil properties at the 5-cm depth (DeBano and Klopatek 1988).

Table 1.—Chemical and physical properties of unburned mineral soils.

Cover	Organic C	Total N	Total P	pH	Texture Sand	Silt	Clay
	%	- - - g/kg - - -			- - - - % - - -		
Pinyon	24.6	1.28	0.47	7.8	50.9	26.7	22.4
Juniper	28.7	2.16	0.50	8.1	49.2	24.6	26.2
Interspace	6.8	0.55	0.44	8.2	45.8	24.2	30.4

Mycorrhizal Analysis

VAM Bioassay

Soils from each microcosm were bioassayed to determine the effects of fire on VAM propagules. Sudan grass [*Sorghum bicolor* (L.) Moench] was used as the host plant. Test tubes (18 X 150 mm) were filled to a depth of 70 mm with autoclaved sand (121°C for three 2-h intervals) followed by a 50-mm depth of inoculum (soil to be tested), and a 20-mm top layer of autoclaved sand. Germinating grass seeds were placed in the top 20-mm layer and grown in a growth chamber under a temperature regime of 25/15°C (day/night), and a 12-h photoperiod. Sudan grass was grown in a set of three test tubes for each of the 27 microcosms for 25 days. Preliminary testing indicated that at 25 days post germination, the pattern of colonization was such that each point of colonization occurred from an individual propagule, and not from the spread of a preexisting colonization along the roots. The authors, when referring to the word "propagule," are adhering to its strict definition—i.e., "consisting of chlamydospores or azygospores, soil borne vesicles, and mycelium or infected root pieces" (anything that induces a VAM infection) (Daniels and Skipper 1982).

Harvested plants were stained using a modified version of the Phillips and Hayman (1970) method. Fine roots were cut into 50 1-cm portions, scored for the presence or absence of infection points, and data were converted into percentages. A positive identification was observed as either an appressorium formation or the presence of internal VAM hyphae.

Arcsin transformations were made of all percent colonization data prior to statistical analysis. VAM colonization data were compared among treatments and cover types using analysis of variance computed on the means for each microcosm. Significant differences ($p < 0.05$) in colonization among experimental treatments and cover types were isolated using Tukey's honest significant difference measure within the SAS statistical software (SAS Institute, Inc. 1985). Simple correlation analyses of the data were used to test the relationship between VAM colonization and maximum temperatures reached in each treatment. A decrease in VAM colonization was defined as percent colonization of the control minus percent colonization of treatment divided by the percent control colonization (Klopatek et al. 1988).

Bacterial Analysis

Number of ammonium- and nitrate-oxidizing bacteria were estimated using the most-probable-number technique (Alexander 1982, Schmidt and Belser 1982). Serial tenfold dilutions were made up to 10^{-8} with 5 tubes per dilution for each sample taken from the 2-5-cm soil layer of each microcosm at preburn, 1-day postburn, and 90-days postburn. Nitrite and nitrate production were qualitatively determined colorimetrically.

Table 2.—Percentage VAM colonization on Sudan grass grown in soils from under pinyon and juniper canopies and interspaces subjected to different burning treatments. Mean equals the average of three replicate microcosms having three subsamples each; ± SE are in parentheses and represent between plot variability.[1]

Treatment	Pinyon	Juniper	Interspace
Control	41.4 (2.8)a,x	43.7 (1.9)a,x	36.4 (2.5)a,x
Dry burn	21.6 (2.7)a,z	9.0 (3.9)b,z	22.4 (0.9)a,y
Wet burn	32.9 (1.7)b,y	25.6 (1.0)c,y	40.4 (3.4)a,x

[1]*Means in a row having the same first letter (a to c) and in columns having the same second letter (x to z) are not significantly different at the 0.05 level.*

RESULTS

Mycorrhizae

There were no statistical differences in percent VAM colonization among unburned controls for juniper, pinyon, and interspace soils (table 2). Plants grown in dry-burned soils of each cover type experienced the greatest decrease in VAM colonization (table 2). Juniper soils burned when dry were the most severely affected, having an average percent colonization on Sudan grass of only 9.0%. Percent VAM colonization of plants grown in pinyon and interspace burned-over dry soils averaged 21.6% and 22.4%, respectively. Both were also significantly lower than their corresponding controls, although not as low as juniper soils.

Plants grown in pinyon and juniper soils burned when wet demonstrated a significant decrease ($p < 0.05$) in percent colonization compared with their controls but not as large a decrease as for soils burned dry (table 2). Pinyon soils burned when wet had a statistically greater average percent colonization than juniper soils, consistent with the pattern in soils burned when dry. Those plants grown in the interspace soil burned when wet demonstrated a slight increase (though not statistically significant) over the controls.

Nitrifying Bacteria

Although both ammonium- and nitrite-oxidizing bacteria were measured, the total NO_2 oxidizing bacteria

were recorded in numbers too low to be statistically examined (<10^3) and will not be discussed. In the control soils, the number of NH_4-oxidizing bacteria was greatest in the interspaces followed by juniper then pinyon (fig. 1a), which corresponds to our earlier findings (Klopatek and Klopatek 1987).

Each of the microcosms examined immediately after burning showed a statistical decrease in the number of bacteria (figs. 1a, 1b). Significant differences were found relative to the soil moisture content at the time of burning.

Both interspace and juniper dry-burned soils exhibited a larger decrease in bacteria compared with the wet-burned soils; while pinyon had over a 60% loss in the wet burned soils compared with a 33% loss in the dry-burned soil.

After a 90-day incubation period, controls from each cover type demonstrated a decrease in the number of bacteria compared with preburned soils (fig. 1b). Juniper controls had the greatest number of bacteria (59.4 X 10^3/g soil), followed by interspace (47.3 X 10^3/g soil), and pinyon (15 X 10^3/g soil).

Both burn treatments of interspace and juniper soils had a lower number of bacteria compared with the controls. In contrast, both the wet- and dry-burned soils of pinyon showed an increase of 106% and 280%, respectively.

Of the wet-burned soils, juniper had the highest number of bacteria preceded by pinyon and interspace, respectively. In the dry-burned soils interspace decreased sharply with bacterial numbers of only 13 X 10^3 g^{-1} soil compared with pinyon (57 X 10^3 g^{-1} soil) and juniper (32 X 10^3 g^{-1} soil) soils.

The number of bacteria increased in both pinyon and juniper wet- and dry-burned soils compared with those soils of day 1 (figs. 1a, 1b). However, interspace showed a significant decline in both wet- and dry-burned soils.

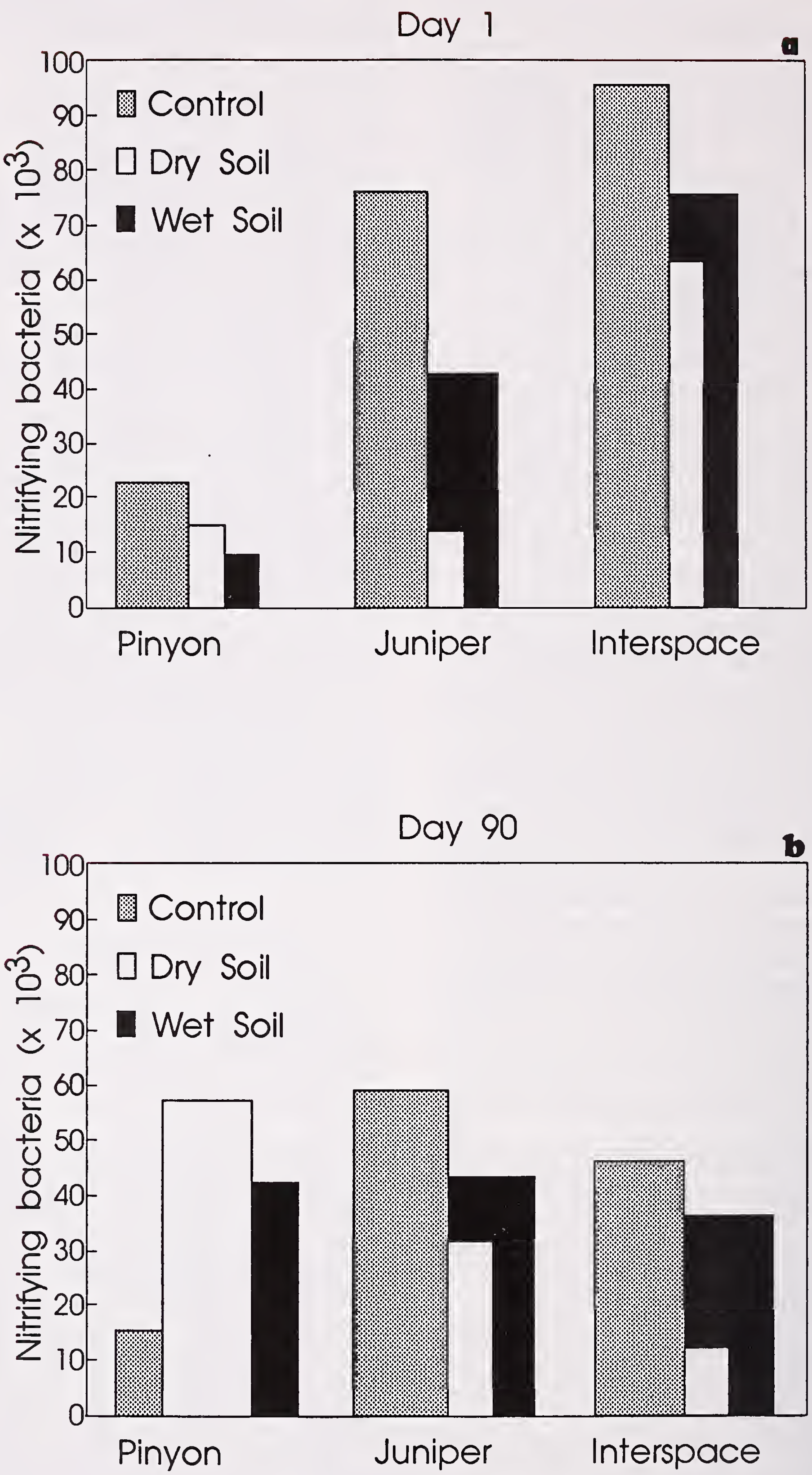

Figure 1.— Number of nitrifying bacteria found (a) immediately after (day 1) and (b) 90-days postburn taken from microcosms containing pinyon, juniper, and interspace soils.

DISCUSSION

Mycorrhizae

There were no statistical differences among unburned controls (table 2). Such uniform colonization was not surprising for the following reasons. Juniper is a VAM-dependent species. Pinyon pine, although ectomycorrhizal, has been reported to have numerous VAM propagules around its base (Klopatek and Klopatek 1987). This is likely due to aeolian deposition of spores. Wind deposits sand particles under pinyon pine (Barth 1980, Klopatek 1987) and probably deposits spores along with the sand. Thus, pinyon may indeed be an important repository for VAM propagules. All grasses in the interspace areas served as hosts for VAM fungi, giving rise to numerous VAM propagules (Klopatek and Klopatek 1987).

Plants grown in dry-burned soils of each cover type experienced the greatest decrease in VAM colonization compared with wet-burned soils, with juniper affected most severely (table 2). Juniper soils experienced the highest temperatures (up to 94°C) (DeBano and Klopatek 1988) which probably resulted in this large loss of VAM. This loss was primarily due to large amounts of litter and duff, in addition to near total combustion of the duff, under these conditions, which contributed to a more intense burn. The higher temperatures were also maintained for a long duration due to smoldering of duff material. Magnitude and duration of heating are the two principal factors causing injury to plants (Hare 1961) and are also likely to be deleterious to VAM fungus propagules.

Plants grown in interspace soils burned when wet had a slight increase (nonstatistical) in VAM colonization compared with the controls. There was little litter and no duff to burn in the interspaces, resulting in a significantly lower ($p < 0.05$) soil temperature than either the pinyon or juniper microcosm when burned wet. This, in combination with the wet soil conditions, minimized any temperature effects on VAM.

A significant correlation ($r^2 = 0.90$, $p < 0.01$) was found between soil temperature and percent decrease in VAM infection (fig. 2). Most soil temperatures ranged from 40° to 60°C, which corresponded to a decrease in VAM colonization of 20% to 50% (fig. 2). VAM colonization was severely reduced when soil temperatures reached 80-90°C, and at 94°C there was a near total loss of VAM colonization (fig. 2).

Fire moderately affected VAM propagules when soil temperatures were less than 50°C. Substantial decreases (>50%) did not occur until soil temperatures reached 60°C. Temperatures of 50° to 60°C resulted in a significant reduction of VAM colonization, probably because a large portion of the infecting mycorrhizal propagules were in the form of infected root and hyphal pieces and were more vulnerable to these temperatures. Hyphae are known to ramify throughout the upper soil horizon, i.e., litter and duff (St. John et al. 1983); once burned or exposed to high temperatures they may be considered to be lost from the VAM propagule population. This loss may contribute to the overall lower VAM population within the pinyon-juniper ecosystem following fire.

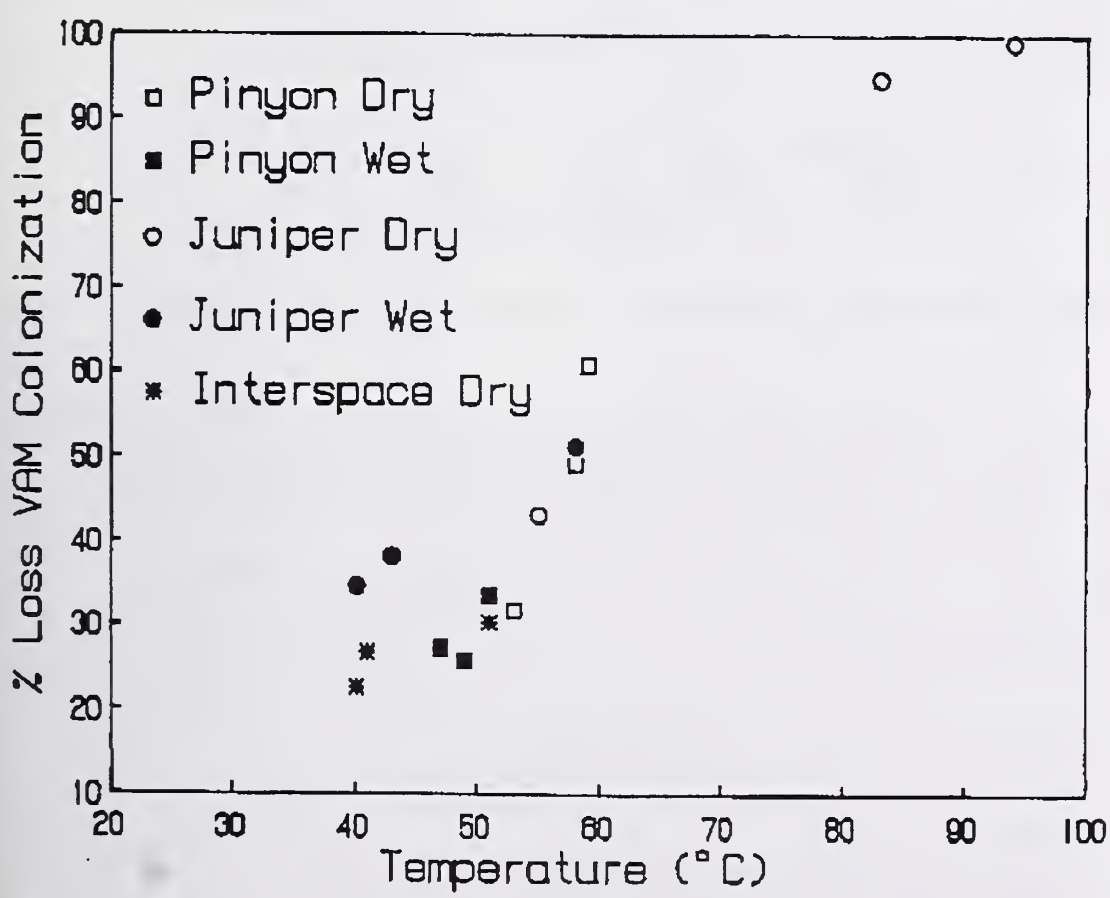

Figure 2.—Percent decrease in VAM colonization as a function of soil temperature. VAM is the average of 3 subsamples from each microcosm, temperature is the average maximum temperatures recorded in each microcosm. $R^2 = 0.90$, $p < 0.001$ from data that were arcsin transformed (see text).

Nitrifying Bacteria

Preburned control interspace and juniper soils exhibited significantly greater bacterial populations than did pinyon soils. This may be due to the presence of allelopathic compounds under the pine canopy, as suggested by Everett[4] for pinyon pine. These compounds have been reported to be water soluble and eas-

[4]*Everett, R., USDA Forest Service, 1986 (personal communication).*

ily volatilized at high temperatures for ponderosa pine soils in New Mexico (White 1986).

Nitrifier populations immediately following the burn (day 1) were dependent on treatment. There was a greater impact on bacteria in the interspace and juniper dry-burned soils, as opposed to the pinyon soils, which were more negatively influenced by the wet-burn treatment. This may be a result of pinyon wet-burned soils maintaining heat for a longer time period than the dry-burned soils.

We found a significant correlation ($r^2 = 0.88$, $p < 0.01$) between nitrifier loss and the resulting burn temperatures (fig. 3). Interspace wet-burned soils had the lowest temperature (31°C) with a corresponding nitrifier loss of 21%; whereas, juniper dry-burned soils had the highest temperature (77°C) and experienced the greatest decrease in nitrifiers of 81%.

At day 90, we found a decrease in nitrifiers in each control microcosm as compared with their preburn counterparts. This correlates with the pattern of utilization of available NH_4 as indicated by the initial increase in nitrification (table 3). At day 90, substrate NH_4 declined (table 3) as did the nitrifying populations. Juniper and interspace soils burned wet and dry maintained a lower level of nitrifying bacteria as compared with the corresponding controls, with the dry burn being lower then the wet. In contrast, there was an increase in nitrifiers in pinyon wet-burned soils and a very significant rise in the dry-burned soils. This increase was presumably due to not only an increase in NH_4 released as a result of the burn, but a possible breakdown of allelopathic compounds found under these trees.

Nitrifying bacteria increased between day 1 to day 90 in both the pinyon and juniper soils burned wet or dry. The initial drop in the bacterial population reflects the direct effect of the fire. The increase in available NH4 (table 3), in combination with a 90-day recovery time, likely generated the increase in the bacterial population. However, interspace soils had a decrease in the control wet- and dry-burned soils when compared with day 1. This supports the above conclusion in that there was not a significant increase of NH_4 in the interspace, and without substrate a downward trend in nitrifiers would be expected.

CONCLUSIONS

At an ecosystem level, fire intensity (increased soil temperatures and duration of these temperatures re-

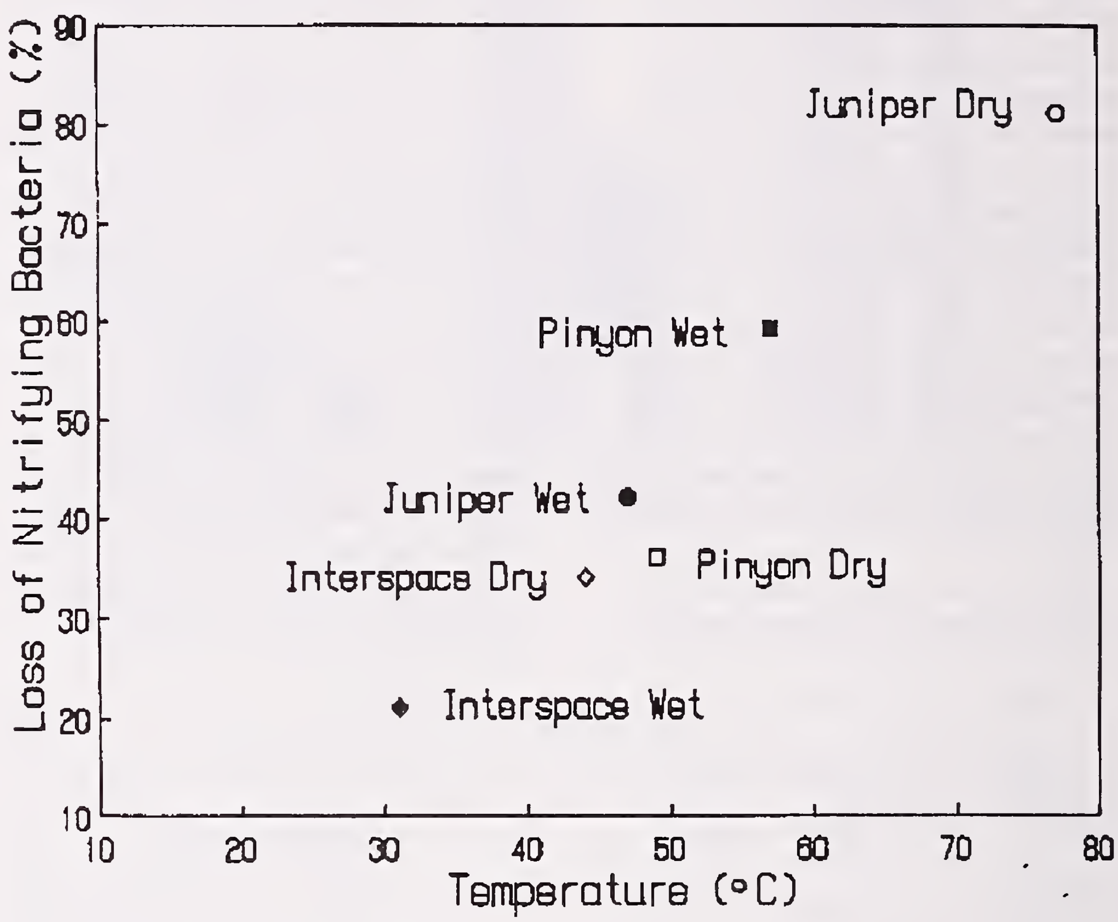

Figure 3.—Percent decrease in nitrifying bacteria as a function of soil temperature.

Table 3.—Prefire and postfire (day 1) concentrations (mg/kg) of NH_4 and NO_3 from under pinyon, juniper, and interspace vegetation.[1]

	NH_4			NO_3		
	Control	Wet	Dry	Control	Wet	Dry
Day 1						
Pinyon	2.0	66.2	125.8	1.7	6.0	5.1
Juniper	2.0	95.2	179.6	4.6	7.1	11.6
Interspace	0.8	6.0	6.3	3.3	5.1	1.5
Day 90						
Pinyon	0.0	11.8	30.3	47.3	40.5	34.2
Juniper	6.7	28.2	31.8	34.5	48.1	61.6
Interspace	6.3	3.2	5.6	6.8	2.0	1.4

[1]*From Klopatek et al., submitted.*

sulting from combustion of organic debris on the soil surface) in combination with soil moisture content at the time of burning, may affect the length of time required for mycorrhizae-dependent plant species to recolonize an area. Allen and Allen (1988), Klopatek and Klopatek (1984), and others have found that the majority of annual plants colonizing a disturbed area are not mycorrhizae dependent.

Plants such as *Salsola kali* (tumbleweed), a pioneer species, are not mycorrhizal; therefore they do not facilitate reestablishment of VAM. These plants can establish themselves because competion by other plants is minimized due to the loss of VAM as well as changes in soil nutrients. Plant recolonization following a fire is, therefore, influenced by VAM availability, in addition to other reported losses such as seed availability (Hare 1961, Martin et al. 1975). Thus, the decrease in VAM may be an important factor in the long-term productivity of all plant species in the pinyon-juniper woodlands.

This study supports what others have found in other ecosystems; namely, burning causes a loss of nitrogen due to nitrification. Disturbance, such as burning, also reduces the other total bacterial and fungal populations, which in turn, can reduce immobilization of nitrogen. Vitousek and Matson (1985) have stated that microbial immobilization is the most important process preventing N losses in a harvested loblolly pine site.

Burning directly causes a reduction of available carbon compounds; this gives the nitrifying population of microbes a competitive edge for nitrogen resources as they do not depend on organic carbon (Alexander 1977). Although the nitrifiers showed significant reductions as a result of the burn treatments, they were not eliminated (fig. 1a). In fact, the population in the canopy-covered areas increased significantly (fig. 1b) as did the NO_3.

As stated earlier, conversion of NH_4 to NO_3 is not an advantageous process because NO_3 is easily lost from the ecosystem.

In conclusion, prescibed burning causes a substantial loss of two important components of the ecosystem, VAM and nitrogen. The ramifications of such losses should be considered when assessing when and if to burn an area.

ACKNOWLEDGEMENTS

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Wettability of an Arizona Chaparral Soil Influenced by Prescribed Burning[1]

John H. Brock and Leonard F. DeBano[2]

Abstract.—Soil samples collected from a chaparral community in central Arizona before and after a prescribed fire were tested for water repellency. Preburn soils were found to have the most severe water repellency in the surface 0-2 cm layer, which decreased with depth. Water repellency was increased in all soil depths by the prescribed fire. Water repellency varied widely within different soil layers indicating that several point samples are necessary for characterizing its vertical and horizontal distribution. General guidelines that managers can use for assessing water repellency are presented.

Arizona chaparral forms a discontinuous belt of vegetation extending northwest to southeast across Arizona. It is dominated by fire-adapted evergreen shrubs that have been classified into several climax vegetation associations (Carmichael et al. 1978). Chaparral has evolved in a climatic regime that favors fire and, as a result, has been exposed to repeated wildfires during its evolution. Because fire has played such an important role in the evolution of chaparral, prescribed fire is used regularly as an accepted management technique. Prescribed fire has been used for improving wildlife habitat and reducing fire hazard and, in conjunction with herbicides, for augmenting water yield (DeBano et al. 1984, Hibbert et al. 1974.).

The continual accumulation of leaf fall on the soil surface increases the organic matter content of the surface mineral soil layers under the shrub canopies. This surface litter contains organic substances that can induce a resistance to water penetration in the underlying mineral soil in unburned stands (Brock and DeBano 1982).

When a plant canopy and intact litter layer are present, soil water repellency has little effect on infiltration. However, when these water-repellent organic materials are volatilized during a fire and are translocated downward in the mineral soil, they can form a severe water repellent layer (DeBano 1981). Thus, in contrast to the preburn condition, this heat-induced water-repellent layer presents problems for the resource manager because infiltration is drastically reduced, surface runoff is increased, and the potential soil erosion greatly increased. This water-repellent soil condition can also prevent microsites from becoming wet during rainstorms, which reduces the chances for successful seed germination and plant establishment during postfire revegetation programs. Although water repellency in unburned stands and its intensification by fire have been reported earlier (DeBano and Krammes 1966, Scholl 1975), no attempt has been made to characterize its distribution both vertically and horizontally in the mineral soil. A three-dimensional characterization of its distribution is important because of its effect on infiltration, which is best modeled as a three-dimensional process. The objective of this study was to describe the three-dimensional distribution of water repellency in chaparral soil both before and after burning. This information can serve as the basis for developing future infiltration models that better describe infiltration into burned chaparral soils.

Methods

The study area was about 2 km southwest of Goodwin, Arizona, a townsite on the Prescott National Forest. Shrub cover on the study area was dominated by shrub live oak *(Quercus turbinella)* and mountainmahogany *(Cercocarpus montanus)*. The soils on this site belong to the Lithic Torriorthent great group and are derived from granite. The surface texture, determined by the hydrometer method (Klute et al. 1986), is a sandy loam containing 54% sand, 36% silt, and 10% clay.

The area was burned by an operational-scale prescribed fire in October 1979. Prior to burning, four 10-meter transects were randomly located on the area to be treated and marked with metal pins. Along each transect, four randomly selected points were marked where two paired plots were located for collecting prefire and postfire soil samples. Each sample plot was 0.25 m^2 and was subdivided into 25 cells, each having an area of 100 cm^2. Plots on the right side of the transect line were designated as preburn sampling plots and those on the left as postburn plots. Soil samples were collected from each cell at 0-2, 2-5, and 5-10 cm depths. This sampling scheme yielded 400 soil samples per soil depth before burn-

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Associate Professor of Environmental Resources, School of Agribusiness and Environmental Resources, Arizona State University, Tempe; and Principal Soil Scientist, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Tempe, AZ 85287-1304.*

ing and another 400 samples after burning.

All soil samples were air-dried and passed through a 2-mm sieve before water repellency was measured by the water drop penetration time method described by DeBano (1981). Water drop penetration times were terminated after 120 seconds had elapsed. Organic matter content was determined by the Walkeley-Black method (Page et al. 1982) on samples, but only for the 0-2 cm surface soil layers before and after burning.

Data were analyzed and descriptive statistics developed using SAS (Statistical Analysis System) computer programs. The experiment was a 2-factor factorial (fire and depth) in randomized complete block design. Transects were considered as blocks and plots were the exponential units. Descriptive statistics were generated across data within plots. Data were subjected to analysis of variance and correlation analyses. Differences among means were tested with the "F" statistic and considered significantly different if $P \leq 0.05$.

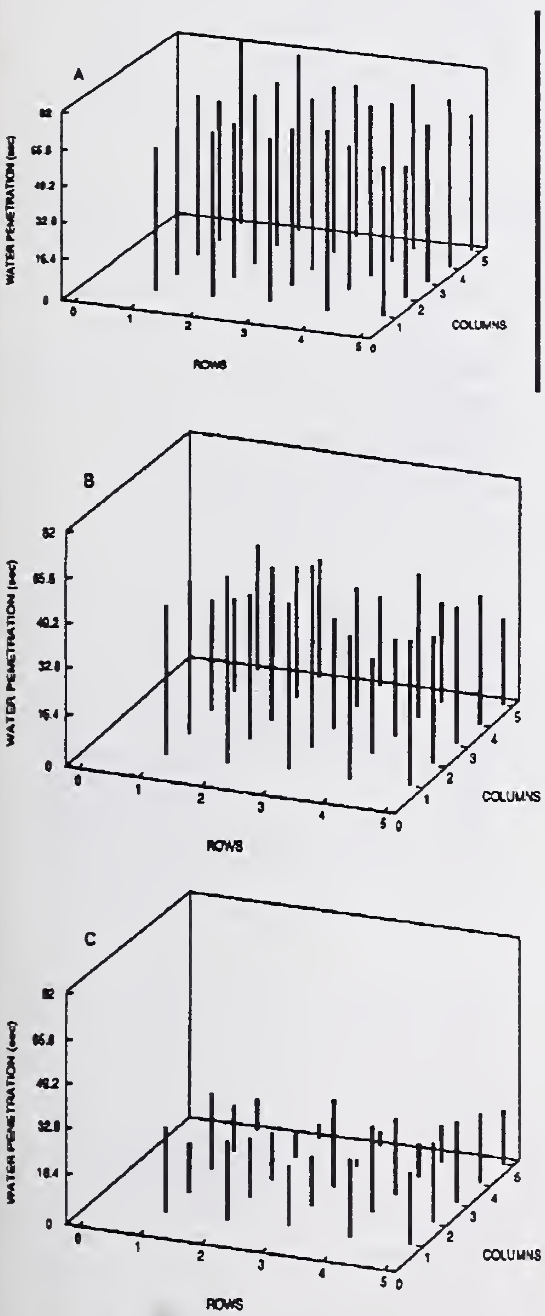

Figure 1.—Average water penetration time (sec) in soil from a chaparral community in central Arizona prior to prescribed burning. A = surface (0-2 cm), B = 2-5 cm depth, and C = 5-10 cm depth.

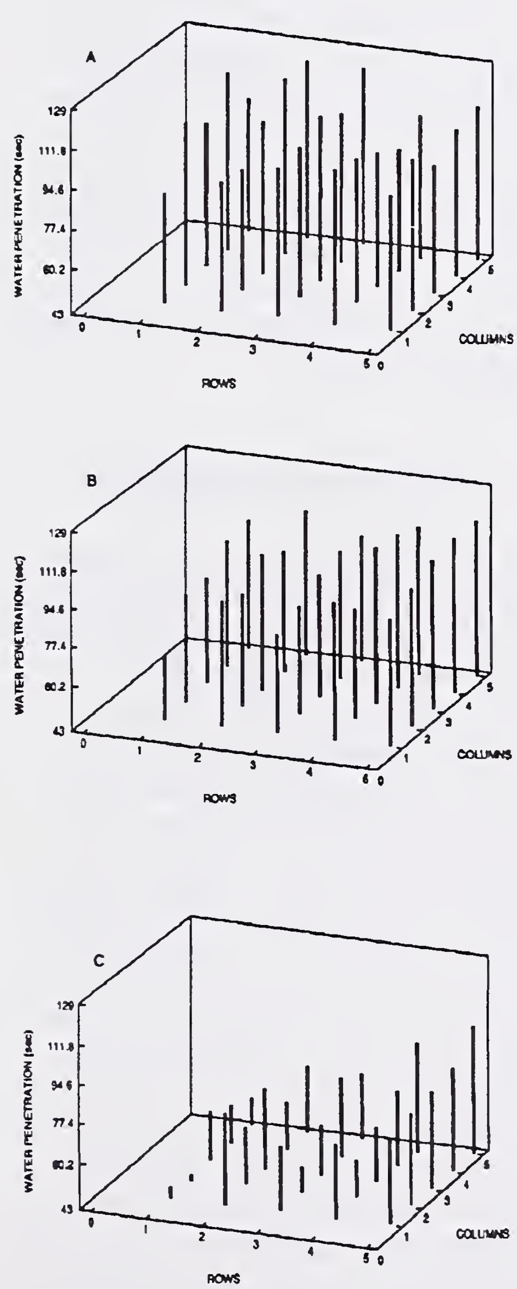

Figure 2.—Average water penetration time (sec) in soil from a chaparral community in central Arizona after prescribed burning. A = surface (0-2 cm), B = 2-5 cm depth, and C = 5-10 cm depth.

Results

Water Repellency

A resistance to wetting was found in all three soil layers prior to burning (fig. 1A, 1B, 1C), although it was most severe in the surface 0-2 cm layer (fig. 1A). The average water drop penetration time of samples from the 0-2 layer was 70.1 sec (± 2.69 SE). The average water drop penetration time in the 2-5 cm layer averaged 47.7 sec (± 2.02 SE), which was 32% less than in the 0-2 cm layer. The average water drop penetration time decreased 71% between the surface and the 5-10 cm layer, where the time was 20.7 sec (± 1.93). The average water drop penetration times were all significantly different at a P = 0.001. A large difference in water repellency was present over the grid surface at each soil depth (fig. 1A, 1B, 1C). For example, the coefficient of variation was 76%, 110%, and 187% in the 0-2, 2-5, and 5-10 cm soil depths, respectively.

The prescribed burn significantly (P = 0.001) increased water repellency in the underlying mineral soils. The surface 0-2 cm layer had an average water drop penetration time of 106.4 sec (± 1.73) (fig. 2A) after burning which was a 151% increase over the time required for water to penetrate soil taken from this same layer prior to burning (fig. 1A).

Average water drop penetration time in the 2-5 cm layer increased 210% over preburn times and was 100.5 sec (± 1.83 SE) (fig. 2B), but was not significantly different (P = 0.05) than in the postburn 0-2 cm layer. Water drop penetration time increased 337% over the preburn times in the 5-10 cm layer and averaged 69.6 sec (± 2.55 SE) after burning (fig. 2C). Less spatial variability was also present among the postburn samples at all soil depths. The coefficients of variation in the 0-2 and 2-5 cm layers were both approximately 30%, whereas the 5-10 cm layer was 73%.

Soil Organic Matter

Burning was found to significantly (P = 0.001) increase organic matter in the surface 0-2 cm layer (fig. 3A, 3B). The average soil organic matter content before burning was 9.0% (± 0.188 SE) and increased to 14.3% (± 0.289 SE) following fire. However, the correlation between mean organic matter and water repellency before and after burning in the surface layer was only 0.09 and was not significant (P = 0.65).

Discussion

The results from this study indicate that water repellency both before and following fire is much the same in Arizona chaparral soils as had been reported earlier in California (DeBano and Krammes 1966). Although water repellency is present in unburned soils it probably does not affect runoff and erosion substantially because it is mitigated by a protective plant and litter cover. Fire increases water repellency by volatilizing organic materials in the litter on the soil surface, and these organic substances move downward into the underlying soil layers where they condense to form a water-repellent layer which is more severe than was present before burning (DeBano 1966).

The increase in both organic matter and water penetration time following fire as measured in this experiment supports this theory. The low correlation between organic matter content and water repellency was caused by the unavoidable incorporation of some unburned organic particles in the surface 0-2 cm layer sample before and after burning. It is nearly impossible to separate ash from the surface mineral soil following burning. The significant increases in water repellency in the 2-5 and 5-10 cm layers following fire further confirm that organic materials are transferred downward in the soil during the combustion of surface litter and plant fuels.

The spatial variations of water repellency both horizontally in a specific layer and vertically between layers indicate that a careful sampling pattern needs to be developed when evaluating postfire water repellency. Single point sampling over depth is not sufficient to adequately characterize the distribution of water repellency in the mineral soil, particularly after burning. The complications that this variable spatial distribution of water repellency has on the development of infiltration models has not yet been resolved. However, it is apparent that infiltration rates based primarily on textural information will have to be reduced substantially following fire. Coarse-textured soils, which normally have high infiltration rates, should receive special attention because they are very sensitive to heat-induced water repellency (DeBano et al. 1970).

Management Implications

Land managers need to be aware of the potential reductions in infiltration that can occur in chaparral soils as a result of burning—either by wildfires or prescribed burns. Water-repellent layers form in much the same way in Arizona chaparral as has been reported in California. The largest problems associated with heat-induced water repellency are in coarse-textured soils containing less than 10% clay. However, on all soils, the loss of the plant canopy and litter layer can lead to increases in raindrop splash and surface erosion. Water repellency on burned areas can be assessed by using the water drop penetration time method. However, a single point sample is usually not sufficient for characterizing the distribution of water-repellent layers because the vertical and horizontal extent of this layer has been found to vary widely. It is recommended that the average of several samples taken in different locations and soil depths be used as the basis for assessing the extent and intensity of fire-induced water repellency.

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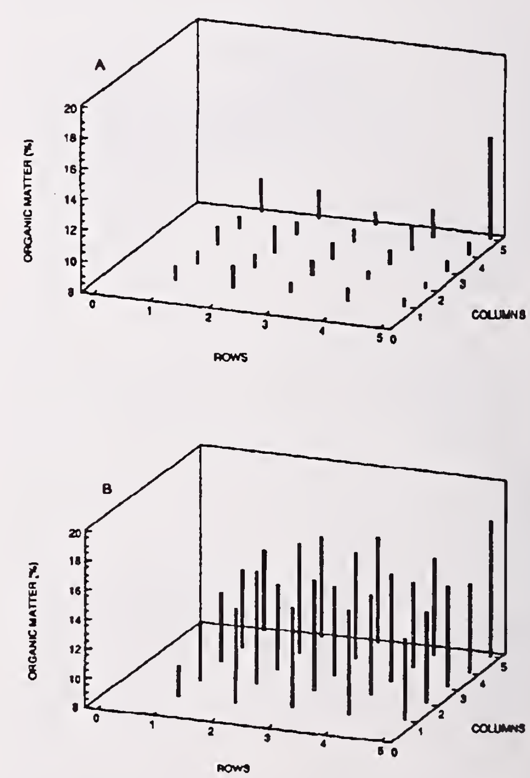

Figure 3.—Average organic matter content in the surface 0-2 cm layer of a chaparral community in central Arizona. A = preburn and B = postburn.

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Fire Effects Information System: An Aid for Fire Use in the Southwest[1]

William C. Fischer and Nancy E. McMurray[2]

Abstract.—The Fire Effects Information System is used to store and retrieve state-of-the-knowledge information regarding the effects of fire on plant species, plant communities, and associated animal species. Essentially, it is a computerized knowledge base of mostly text-type information organized in an encyclopedic fashion. The system was developed to aid prescribed fire planning.

The Fire Effects Information System (FEIS) was developed by the Intermountain Research Station in cooperation with the University of Montana. This system is a computerized information storage and retrieval system that was developed to be an authoritative, easy to access source for information about the effect of fire on individual plant and animal species and on the plant communities in which these species reside.

The expected effect of fire on plant communities is a major consideration affecting decisions to use fire to accomplish a variety of wildland vegetation management objectives. To obtain a specific desired result from a fire treatment, the fire prescription must be based on the best available information and experience regarding the response of target plant species to fire and how this response varies according to such factors as fire severity, season, phenological state, successional status, site characteristics, and other biological and environmental considerations. Many managers perceive a lack of such fire effects information as a barrier to the effective use of prescribed fire for vegetation management (Kickert et al. 1976, Kilgore and Curtis 1987, Noste and Brown 1981, Taylor et al. 1975). However, a substantial body of information exists about fire effects generally and plant response to fire in particular, especially for the species of primary management concern. The problem, largely one of the accessibility of such information, has two facets: (1) there is no single "best" route to the available information, and (2) the information is generally unorganized and uninterpreted for the purpose of aiding fire management decisions. The Fire Effects Information System is a unique solution to this problem.

What It Is and What It Is Not

The Fire Effects Information System is a computerized knowledge management system that stores and retrieves state-of-the-knowledge, English-language texual information organized in an encyclopedic fashion. It is unlike most information systems available to natural resource managers. It is not a computerized bibliography although a computerized bibliography is an important appendage to the system. It is not a numerical data base although the system does accommodate numerical data. And the information provided by the system is ready to use; it does not have to be decoded.

For those abreast of computer science trends, FEIS is an object-oriented, frame-based, knowledge-based system implemented in the LISP programming language. FEIS was developed using concepts, methods, and techniques from the rapidly expanding field of artificial intelligence (AI), but it is not an expert system. (For details on the design and structure of FEIS and development of its software, see Fischer and Wright 1987).

The Fire Effects Information System consists of three components: the knowledge base, the query program, and the builder program. The knowledge base contains the fire effects and related information that is available to users of the system. The query program allows access to the knowledge base but does not allow any changes. It is designed for people who are unskilled in computer use. The builder program is used by those who are adding to or editing the knowledge base. The user of the builder program is expected to be familiar with the structure of the knowledge base and is expected to be skilled in computer use. Because it is the object of the system, the knowedge base is described in more detail below.

The Knowledge Base

The FEIS knowledge base is designed to accept information in three major categories: plant species, ecosystems, and wildlife species. The ecosystem category includes three levels of classification: an ecosystem

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[2]William C. Fischer is a Research Forester, Intermountain Research Station, Forest Service, U.S. Department of Agriculture, Ogden UT, stationed at Intermountain Fire Sciences Laboratory, P.O. Box 8089, Missoula, MT; and Nancy E. McMurray is an Ecologist, Systems for Environmental Management, P.O. Box 8868, Missoula, MT.

level, a cover type level, and a habitat type or plant community level. For each category and level, the knowledge base contains state-of-knowledge textual information for various predetermined topics for several subject matter areas. Topics by subject matter area for each of the three categories of information are listed in table 1. The knowledge base will accept information only for the predetermined topics listed in table 1. Addition of other topics is relatively simple for someone who is familiar with the structure of the system and capable of programming in the LISP language. A topic title will not appear on the screen of the user's computer terminal until an entry for that topic exists in the knowledge base. Fischer (1987) and Fischer and Wright (1987) provide examples of FEIS output essentially as it would be displayed on the screen of a user's computer terminal.

Knowledge Base Development

The information contained in the FEIS knowledge base is the product of a rigorous process that includes (1) making a thorough bibliographic search to identify literature related to the topics listed in table 1, (2) obtaining hard copy of all such literature, (3) reading the literature, evaluating its reliability, and summarizing useful information, (4) resolving conflicts, if possible, between contradictory information, (5) synthesizing fire effects information, and (6) entering the information into the knowledge base. The fire research team responsible for knowledge base development consists of professional biologists trained in the areas of botany, wildlife biology, range science, and forestry. On the average, it takes about 10 days for a team member to complete an initial species or ecosystem writeup and enter it into the knowledge base. Following the entry, the information for a given species or ecosystem is reproduced on paper and sent for technical review to scientists, staff specialists, and managers who have expert knowledge of the species or ecosystem. The information in the knowledge base is revised, as necessary, to reflect this technical review. Information in the knowledge base is periodically revised to incorporate knowledge from more current literature.

Content of the Knowledge Base

As of January 1, 1989, information for 176 plant species (25 trees, 70 shrubs, 64 grasses, and 17 forbs), 13 wildlife species, and 1 ecosystem (including 8 cover types) was contained in the FEIS knowledge base (see appendix A). The distribution of species according to their occurrence in Forest-Range Environmental Study ecosystems (Garrison et al. 1977) is presented as appendix B. The Bureau of Land Management (BLM) and the National Park Service (NPS), U.S. Department of the Interior (USDI) have been the primary sponsors for the development of a prototype knowledge base. A majority of the species included in the present knowledge base are, consequently, those common to the semi-arid Western rangelands managed by the BLM and the ponderosa pine forests and plains grasslands of Wind Cave National Park, South Dakota. The NPS designated Wind Cave National Park as a prototype park for knowledge base development. Currently, the BLM is sponsoring knowledge base additions of species that occur in the chaparral-mountain shrub, desert shrub, pinyon-juniper, and sagebrush ecosystems. The NPS is currently sponsoring the addition of species that occur in Yellowstone National Park.

Current Access to The System

FEIS was developed on a Digital Equipment Corporation (DEC) VAX 750 computer at the University of Montana and is now on a Data General (DG) MV 4000 computer at the Intermountain Fire Sciences Laboratory in Missoula, MT.[3] The system also resides on a BLM DG MV10000 computer at the Boise Interagency Fire Center, Boise, ID. BLM personnel access the system using IBM-compatible personal computer (PC), a 1200 baud phone modem, and terminal emulation/communications software that can emulate a DG 400 terminal. Forest Service personnel, at sites where TELNET is installed, have been allowed DG access to the system at the Fire Sciences Laboratory via the deflected drawer process. Additionally, the system has been delivered to the NPS Branch of Fire Management at Boise for installation on its DEC VAX 750.

The BLM and NPS systems contain only the knowledge base and query components. The builder program resides only at the Fire Sciences Laboratory and is presently restricted to use by fire research personnel involved in knowledge base development. A personal computer (PC) version of the FEIS query program and knowledge base is available at Wind Cave National Park for operational evaluation. A PC builder program is not yet available.

Planning for a widely available operational implementation of the Fire Effects Information System is under way. Feasible alternatives have been identified for consideration by potential sponsors. Initial action is aimed toward some form of interagency implementation that would allow access by all potentially interested users. Widespread, multiagency operational implementation of FEIS is probably at least 2 years in the future.

[3]The use of trade and company names is for the benefit of the reader; such use does not constitute an official endorsement or approval of any service or product by the U.S. Department of Agriculture to the exclusion of others that may be suitable.

Table 1—Information by category, subject matter area, and topic contained in the FEIS knowledge base.

Plant Species Category

Species name
Abbreviation
Synonyms
Common names
Taxonomy
Life form
Compiled by and date
Last revised by and date
References

DISTRIBUTION & OCCURRENCE

General distribution
Ecosystems
States
Administrative units
BLM physiographic regions
Kuchler plant associations
SAF cover types
Habitat types
References

VALUE AND USE

Wood products value
Importance to livestock & wildlife
Palatability
Food value
Cover value
Value for rehabilitation of disturbed sites
Other uses and values
Management considerations
References

BOTANICAL & ECOLOGICAL CHARACTERISTICS

General botanical characteristics
Growth form
Raunkiaer life form
Grime plant strategy class
Grime regenerative strategy class
Regeneration processes
Site characteristics
Successional status
Seasonal development
References

PLANT ADAPTATIONS TO FIRE

General adaptations to fire
Lyon-Stickney survival strategy
Noble-Slatyer vital attributes
Rowe mode of persistence
References

FIRE EFFECTS

Fire effects on plant
Discussion & qualification
Plant response to fire
Discussion and qualification
References

FIRE CASE STUDY

Case study name
Reference
Season-severity class
Study location
Preburn vegetation
Target species phenological site
Site description
Fire description
Fire effects on target species
Fire management implications

Wildlife Species Category

Species name
Abbreviation
Common name
Taxonomy
Order
Class
Compiled by and date
Last revised by and date
References

DISTRIBUTION & OCCURRENCE

General distribution
Ecosystems
States
Administrative units
BLM physiographic regions
Kuchler plant associations
SAF cover types
Plant communities
References

BIOLOGICAL DATA & HABITAT REQUIREMENTS

Timing of major life history events
Preferred habitat
Cover requirements
Food habits
Predators
References

FIRE EFFECTS & USE

Direct fire effects on animal
Habitat related fire effects
Fire use
References

Ecosystems Category

ECOSYSTEM LEVEL

Ecosystem name
Compiled by and date
Last revised by and date
Classification key
FRES number
Kuchler vegetation types
Ecosystem distribution
References

PRODUCTIVITY

Characteristics/productivity classes
Dominant species/productivity classes
Potential production
References

CONDITION & TREND

Characteristics of condition classes
Indicators of trend
Qualification & discussion
References

FIRE ECOLOGY & EFFECTS

Fuels, flammability, & fire occurrence
General fire effects
References

Ecosystems Category (cont.)

COVER TYPE LEVEL

Cover type
Compiled by and date
Last revised by and date
Abbreviation
Classification key
Distribution
Site characteristics
Vegetative composition
Successional trends
References

VALUE & USE

Wood products
Livestock range
Wildlife habitat
Other values and uses
References

FIRE ECOLOGY & EFFECTS

Fuels, flammability, & fire occurrence
Immediate fire effects on site
Initial vegetative response
Long term vegetative response
Fire effects on grazing potential
Fire effects on wildlife habitat & populations
Fire use potential
Rehabilitation of burned sites
Fire management considerations

PLANT COMMUNITY LEVEL

Community or group name
Abbreviation
Description
Community type composition
Distribution & occurrence
Site characteristics
Vegetative composition
Productivity
Successional trends

MANAGEMENT CONSIDERATIONS

Wood products
Livestock range
Wildlife habitat
Other considerations
References

FIRE EFFECTS

Fuels, flammability, & fire occurrence
Initial community response
Long term community response
Fire effects on grazing potential
Fire effects on wildlife habitat and populations
Fire use potential
References
Fire case studies

Relevance of FEIS to The Southwest

Most Southwestern ecosystems are represented by the plant species presently contained in the FEIS knowledge base (see appendix B). Perhaps as many as 70 percent of the species contained in the knowledge base occur in these Southwestern ecosystems. In addition, many of the species scheduled for inclusion in the knowledge base during 1989 occur in the desert shrub, chaparral-mountain shrub, pinyon-juniper, and Southwestern shrubsteppe ecosystems of this area. Consequently, FEIS is a potential resource for Southwestern fire managers and resource specialists involved in planning fire use and evaluating wildfire effects. While BLM personnel have ready access to FEIS, provided they have the required computer hardware and software, other USDI agencies may be able to arrange similiar access. Forest Service access is presently limited but could be improved by temporary installation of FEIS on Regional DG systems.

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Appendix A: Species and cover types contained in FEIS knowledge base (Jan. 1, 1989)

Tree Species

Acer negundo, boxelder
Celtis occidentalis, hackberry
Cercocarpus ledifolius, curlleaf mountain-mahogany
Cercocarpus montanus, true mountain-mahogany
Cowania mexicana ssp *stansburiana*, Stansbury cliffrose
Fraxinus pennsylvanica, green ash
Juniperus occidentalis, western juniper
Juniperus osteosperma, Utah juniper
Juniperus scopulorum, Rocky Mountain juniper
Pinus albicaulis, whitebark pine
Pinus aristata, Rocky Mountain bristlecone pine
Pinus balfouriana, foxtail pine
Pinus edulis, pinyon
Pinus flexilis, limber pine
Pinus longaeva, Great Basin bristlecone pine
Pinus monophylla, singleleaf pinyon
Pinus ponderosa var *scopulorum*, Interior or Black Hills ponderosa pine
Populus tremuloides, aspen
Prunus americana, American plum
Prunus pensylvanica, pin cherry
Prunus virginiana, chokecherry
Quercus gambellii, Gambel oak
Quercus macrocarpa, bur oak
Quercus turbinella, turbinella oak
Rhus glabra, smooth sumac

Shrub Species

Ambrosia (Franseria) deltoidea, triangle bursage
Ambrosia (Franseria) dumosa, white bursage
Amelanchier alnifolia, Saskatoon serviceberry
Amelanchier utahensis, Utah serviceberry
Amorpha canescens, leadplant
Arctostaphylos pungens, pointleaf (Mexican) manzanita
Artemisia abrotanum, oldman wormwood
Artemisia arbuscula ssp *arbuscula*, gray low sagebrush
Artemisia arbuscula ssp *thermopola*, hotsprings sagebrush
Artemisia argillosa, coaltown sagebrush
Artemisia bigelovii, Bigelow sagebrush
Artemisia cana ssp *viscidula*, mountain silver sagebrush
Artemisia cana ssp *bolanderi*, Bolander silver sagebrush
Artemisia cana ssp *cana*, plains silver sagebrush

Artemisia filifolia, sandsage or sand sagebrush
Artemisia frigida, fringed sagebrush
Artemisia longiloba, early or alkali sagebrush
Artemisia nova, black sagebrush
Artemisia papposa, fuzzy sagebrush
Artemisia pedatifida, birdfoot sagebrush
Artemisia pygmaea, pygmy sagebrush
Artemisia rigida, stiff or scabland sagebrush
Artemisia spinescens, budsage or bud sagebrush
Artemisia tridentata ssp *tridentata*, basin big sagebrush
Artemisia tridentata ssp *vaseyana*, mountain big sagebrush
Artemisia tridentata ssp *wyomingensis*, Wyoming big sagebrush
Artemisia tripartita ssp *rupicola*, Wyoming threetip sagebrush
Artemisia tripartita ssp *tripartita*, tall threetip sagebrush
Atriplex canescens, four-wing saltbrush
Atriplex confertifolia, shadscale
Atriplex gardneri, saltsage
Ceratoides lanata, winterfat
Chrysothamnus nauseosus, grey rabbitbrush
Chrysothamnus viscidiflorus, green rabbitbrush
Coleogyne ramosissima, blackbrush
Ephedra nevadensis, Nevada ephedra
Ephedra viridis, green ephedra
Fallagia parodoxa, Apache plume
Flourensia cernua, tarbush
Grayia brandegei, spineless hopsage
Grayia spinosa, spiny hopsage
Gutierrezia sarothrae, broom snakeweed
Holidiscus discolor, oceanspray
Holidiscus dumosus, bush oceanspray
Larrea tridentata, creosotebush
Leplodactylon pingens, prickly phlox
Opuntia polyacantha, plains prickly pear
Potentilla fruticosa, shrubby cinquefoil
Prunus andersoni, desert peach
Purshia glandulosa, desert bitterbrush
Purshia tridentata, antelope bitterbrush
Rhus aromatica, fragrant sumac
Rhus trilobata, skunkbrush sumac
Ribes americanum, American black currant
Ribes aureum, golden currant
Ribes cereum, wax currant
Ribes lacustre, swamp currant
Ribes montigenum, gooseberry currant
Ribes odoratum, buffalo currant
Ribes setosum, bristley currant
Ribes velutinum, desert gooseberry
Sarcobatus baileyi, Bailey greasewood
Sarcobatus vermiculatus, black greasewood
Symphoricarpos longiflorus, Longfellow snowberry
Symphoricarpos oreophilus, mountain snowberry
Tetradymia canescens, spineless horsebrush
Tetradymia glabrata, littleleaf horsebrush
Tetradymia nuttallii, Nuttall horsebrush
Tetradymia spinosa, spiny horsebrush
Toxicodendron rydbergii, poison ivy

Graminoid Species

Agropyron cristatum (A. pectiniforme), fairway wheatgrass
Agropyron desertorum, standard wheatgrass
Andropogon barbinodis, cane bluestem
Andropogon gerardii, big bluestem
Andropogon halli, sand bluestem
Aristida purpurea (A. longiseta), three-awn grass
Bouteloua curtipendula, sideoats grama
Bouteloua eriopoda, black grama
Bouteloua gracilis, blue grama
Bouteloua hirsuta, hairy grama
Bromus carinatus, California brome
Bromus inermis, smooth brome
Bromus japonicus, Japanese brome
Bromus marginatus, mountain brome
Bromus mollis, soft chess
Bromus rubens, red brome
Bromus tectorum, cheatgrass or downy brome
Buchloe dactyloides, buffalograss
Calomovilfa longifolia, prairie sandreed
Carex heliophila, sun sedge
Danthonia intermedia, timber oatgrass
Danthonia spicata, poverty oatgrass
Danthonia unispicata, onespike oatgrass
Elymus canadensis, Canada wildrye
Elymus elymoides, (Sitanion hystrix), bottlebrush squirreltail
Elymus glaucus, (E. virescens), blue wildrye
Elymus lanceolatus, (Agropyron dasystachyum, A. elmeri, A. riparium), thickspike wheatgrass
Festuca idahoensis, Idaho fescue
Festuca scabrella, rough fescue
Festuca thurberi, Thurber fescue
Hilaria belangeri, curly mesquite
Hilaria jamesii, galleta
Hilaria mutica, tobosa
Hilaria rigida, big galleta
Koeleria cristata, prairie junegrass
Leucopoa kingii, spike fescue
Leymus (Elymus) ambiguus, Colorado wildrye
Leymus (Elymus) cinereus, basin wildrye
Leymus (Elymus) salinus, Salina wildrye
Muhlenbergia cuspidata, plains muhly, Stonyhills muhly
Muhlenbergia racemosa, green muhly
Muhlenbergia richardsonis, mat muhly
Oryzopsis hymenoides, Indian ricegrass
Pascopyrum (Agropyron) smithii, western wheatgrass
Poa arida, plains bluegrass
Poa cusickii, Cusick bluegrass
Poa fendleriana, mutton bluegrass
Poa secunda, (P. ampla, P. canbyi, P. juncifolia, P. nevadensis, P. sandbergii), Sandberg bluegrass
Psathyrostachys juncea (Elymus junceus), Russian wildrye
Pseudoroegneria spicata (Agropyron spicatum, A. inerme), bluebunch wheatgrass
Schizachyrium (Andropogon) scoparius, little bluestem
Sporobolus airoides, alkali sacaton
Sporobolus asper, tall dropseed
Sporobolus cryptandrus, sand dropseed
Sporobolus flexuosus, mesa dropseed
Stipa columbiana, Columbia needlegrass
Stipa comata, needle-and-thread grass

Stipa lettermanii, Letterman needlegrass
Stipa thurberiana, Thurber needlegrass
Stipa viridula, green needlegrass
Taeniatherum caput-medusae, medusahead
Vulpia (Festuca) microstachys, small fescue
Vulpia myuros (Festuca megalura), foxtail fescue
Vulpia (Festuca) octoflora, six-weeks fescue

Forb Species

Achillea millefolium, common yarrow
Artemisia camprestris, sagewort wormwood, western sagebrush
Artemisia dracunculus, tarragon
Artemisia ludoviciania, Louisiana sagewort
Balsamorhiza hookeri, hairy balsamroot
Balsamorhiza sagitata, arrowleaf balsamroot
Centaurea diffusa, tumble knapweed
Centaurea maculosa, spotted knapweed
Centaurea solstitialis, yellow starthistle
Descurainia pinnata, tansymustard
Descurainia sophia, flixweed tansymustard
Potentilla glandulosa, sticky cinquefoil
Potentilla hippiana, horse cinquefoil
Potentilla newberryi, Newberry cinquefoil
Ranunculus glaberrimus, sagebrush buttercup
Sisymbrium altissimum, tumble mustard
Sisymbrium linifolium, flaxleaf plains mustard

Cover Types

Artemisia arbuscula ssp *arbuscula* C.T., low sagebrush cover type
Artemisia arbuscula ssp *thermopola* C.T., hotsprings sagebrush cover type
Artemisia cana ssp *bolanderi* C.T., Bolander silver sagebrush cover type
Artemisia cana ssp *cana* C.T., plains silver sagebrush cover type
Artemisia cana ssp *viscidula* C.T., mountain silver sagebrush cover type
Artemisia filifolia C.T., sand sagebrush cover type
Artemisia frigida C.T., fringed sagebrush cover type
Artemisia nova C.T., black sagebrush cover type

Wildlife Species

Amphibians & Reptiles

Ambystoma macrodactylum ssp *krausei*, northern long-toed salamander
Crotalus viridis, western rattlesnake
Sceloporus graciosus, sagebrush lizard
Scophiopus intermontanus, Great Basin spadefoot toad

Birds

Aquila chrysaetos, golden eagle
Athene cunicularia, burrowing owl
Buteo regalis, ferruginous hawk
Centrocercus urophasianus, sage grouse
Falco mexicanus, praire falcon

Mammals

Antilocapra americana, pronghorn antelope
Lepus californicus, black-tailed jack rabbit
Perognathus parvus, Great Basin pocket mouse
Spermophilus townsendii, Townsend's ground squirrel

Appendix B:

Number of plant species by ecosystem represented in FEIS knowledge base

Ecosystem	Trees	Shrubs	Graminoids	Forbs	Total
Forest & Woodland Ecosystems					
White-red-jack pine	1	1	-	-	2
Spruce-fir	1	1	-	-	2
Longleaf-slash pine	-	-	-	-	0
Loblolly-shortleaf pine	-	-	-	-	0
Oak-pine	2	-	-	-	2
Oak-hickory	7	9	14	-	30
Oak-gum-cypress	3	-	-	-	3
Elm-ash-cottonwood	8	17	25	2	52
Maple-beech-birch	4	2	1	4	11
Aspen-birch	5	-	2	-	7
Douglas-fir	11	21	30	8	70
Ponderosa pine	21	43	50	11	125
Western white pine	-	3	-	-	3
Fir-spruce	11	18	25	7	61
Hemlock-Sitka spruce	-	4	-	-	4
Larch	-	2	-	-	2
Lodgepole pine	7	7	3	1	18
Redwood	-	1	-	-	1
Western hardwoods	5	6	3	1	15
Shrubland Ecosystems					
Sagebrush	17	56	52	13	138
Desert shrub	7	44	30	6	87

Appendix B. (continued)

Ecosystem	Trees	Shrubs	Graminoids	Forbs	Total
Shinnery	-	2	2	-	4
Texas savanna	-	1	7	-	8
Southwestern shrubsteppe	-	3	8	1	12
Chaparral-mountain shrub	15	38	46	5	104
Pinyon-juniper	16	50	49	10	125
Grassland Ecosystems					
Mountain grasslands	10	37	45	10	102
Mountain meadows	-	-	-	-	0
Plains grasslands	12	27	45	9	93
Prairie	8	13	28	4	53
Desert grasslands	5	35	26	5	71
Wet grasslands	-	-	3	-	3
Annual grasslands	-	-	1	-	1
Alpine Ecosystems					
Alpine	1	6	6	3	16

FIREMAP: Simulation of Fire Behavior — A GIS Supported System[1]

Maria J. Vasconcelos, D. Phillip Guertin, Malcolm J. Zwolinski[2]

Abstract—The purpose of the FIREMAP system is to simulate fire behavior in a spatially nonuniform environment, given a set of environmental conditions. This system integrates a fire behavior prediction system (BEHAVE) and a Geographic Information System (Map Analysis Package) in a framework that allows simulation of the actual spread of a fire over a digital elevation model. An example illustrating the potential usefulness of this simulation tool is given.

Forest management and planning is a multiobjective task. Fire will impact the potential resource outputs and is one of the many factors that should be addressed, depending on context and goals. From a planning viewpoint, we must be able to predict the consequences of site-specific management actions on the fire characteristics (areas burned and intensity) for the area of concern.

From the literature, fire behavior prediction and modeling are limited to homogeneous cover types (Rothermel 1972,1983), where the other driving variables (weather and topography) are assumed to be uniform over the same area. These models are, therefore, unsuitable for dealing with the different spatial combinations of vegetation which a management alternative might consider and cannot be readily applied to patchy enviroments. The FIREMAP system attempts to address the problem of spatial/temporal variability of the driving variables which has not yet been adequatelly considered in fire modeling.

The FIREMAP System

The objective of FIREMAP is to simulate the consequences of hypothesized changes in vegetation composition and density on the fire characteristics (area burned and fire intensity) in well known ecosystems. This system estimates fire characteristics taking spatial and temporal variability into account and simulating the spread of fire in discrete time steps.

FIREMAP is mainly applicable as a prescriptive tool but it can also be used for predicting fire behavior in "on site" situations when time effects have to be analyzed. Mapped outputs can also provide a basis for better communication.

Model Development Tools

The development of this simulation tool consists of integrating a fire behavior modeling system - DIRECT, a module from the BEHAVE system (Andrews 1986), with a Geographic Information System—MAP (Tomlin 1985).

DIRECT uses Rothermel's fire spread model (1972) to predict fire characteristics for a given continuous and relatively homogeneous area. Andrews (1980) reports favorable statistical comparisons between model predicted and observed rates of fire spread when burning conditions are uniform. To deal with spatial nonuniformity of fuels, weather and topography the field must first be partitioned into homogeneous units. This partitioning allows the use of the spread model within each unit (Fujioka 1985).

MAP is a raster based GIS designed to run on IBM compatible microcomputers. It has a grid cell data structure in which map information is stored as numeric values in arrays, each cell representing an uniform parcel of land located within the overall rectangular grid. MAP provides for storage, processing and display of cartographic data allowing for input in the form of grid cells, digitized points, lines or polygons. The processing capabilities consist of spatial data base management, spatial statistics and cartographic modeling that use sequential processing of mathematical operations plus maps and a common database to store intermediate results (Berry and Reed 1987). The simulation of fire spread is based on the "distance function" spread. This distance accumulating process can be limited to upward or downward directions over a 3-dimensional surface (Tomlin 1985).

FIREMAP Structure and Interface

There are three main sections in FIREMAP. The first section generates the input overlays required to run the fire model which are based on meteorological data, time of the day, month of the year, an altitude overlay, a vegetation overlay, and a stream channels overlay, by following the framework described in Rothermel (1983). The second section consists of a program, written in FOR-

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Division of Forest-Watershed Resources, School of Renewable Natural Resources, University of Arizona.*

TRAN77, that reads the maps as arrays, runs the fire model and creates the output overlays that store the values describing fire characteristics for each cell in the data base (fig.1).

The third section consists of the actual simulation of the fire spread for the given conditions. A source (or sources) of ignition, is selected. It is assumed that the wind is consistently blowing from the same direction, and an overlay with a constant inclination in the direction of the wind is created (WIND).

The spread operation previously described, is used over this surface, through an overlay, FRICTION, that has assigned to each grid cell the number of time units it takes the cell to be consumed by a fire, with the given conditions. This calculation is based on the rate of spread (feet/min) and size of each cell (feet). The value of friction assigned to stream channels is the result of a calibration done for this particular situation. Using spread operation in the direction of the wind, fire spreads preferentially through the path of least resistance, or the cells taking the least time to be consumed by the same fire, up to the point where the predefined simulation time is reached.

The other output overlays (heat per unit area, fireline intensity, effective windspeed, flame length, reaction intensity and direction of maximum spread) give useful mapped information about the characteristics of the fire in each of the grid cell units, for the time interval on which the weather conditions utilized apply. Because the fire model predicts characteristics of the fire in the flaming front, they are valid only when the fire is still burning on that cell. Three-dimensional displays of the areas burning can be included.

Application of Firemap

Problem Description

The area considered in this example is located in the Spotted Mountain area, on the Fort Apache Indian Reservation in east-central Arizona. It corresponds to an area 9 square miles in size with elevations ranging from 5800 to 7000 feet, on steep slopes. The vegetative cover consists of three types: ponderosa pine stands *(Pinus ponderosa)* with variable crown and understory densities, pine-Douglas fir stands *(Pinus ponderosa-Pseudotsuga menziesii)*, and pinyon-juniper. The vegetation management alternatives considered are no intervention and harvesting by selective cutting (fig. 2).

To analyze the influence of one variable in the system, all other variables have to remain constant. Here spatial arrangement of vegetation, under harvest or non-harvest conditions is the driving variable under consideration; therefore, weather conditions and source of the fire are the same for the two simulations.

The weather data utilized were taken at Ivins Canyon on June 11, 1988 at 3:00 p.m.: Dry bulb temperature - 82° F; Relative humidity - 12%; Windspeed at 6ft - 12 m.p.h.; Wind direction - South. Simulations are performed for a period of three hours, where each time step is one hour long. Another example is given on how a change in wind direction from south to west, at the end of the second hour would affect the area burned.

Results

The final result of the simulation is shown on figure 3. Figure 4 shows how a change in the wind direction (accounted for between time steps) can affect the area and shape of the burned area, illustrating the utilization of FIREMAP for a known weather situation. Figure 5 displays the expected flame lengths for the no intervention alternative during the three hours of simulation.

Conclusions

The results of the FIREMAP application described here indicate that the approach followed works reasonably well. The integration of a fire model like BEHAVE with a GIS may be an efficient way of accounting for spatial variability when attempting to predict fire behavior.

FIREMAP, in the same way as BEHAVE, is a direct implementation

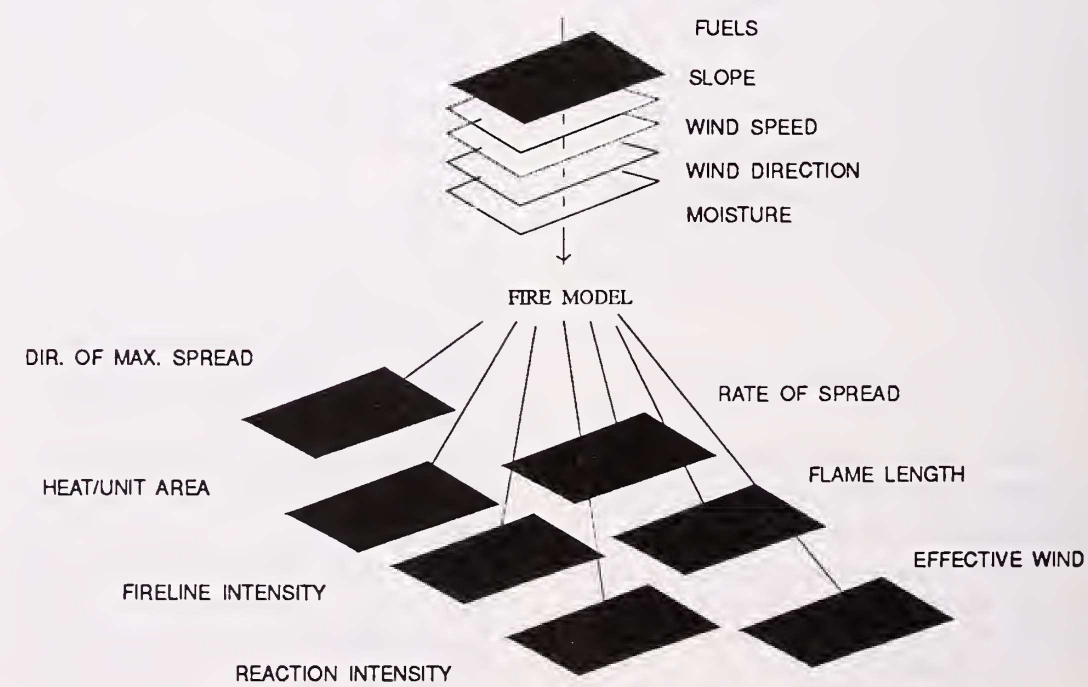

Figure 1.—Using GIS as a data base.

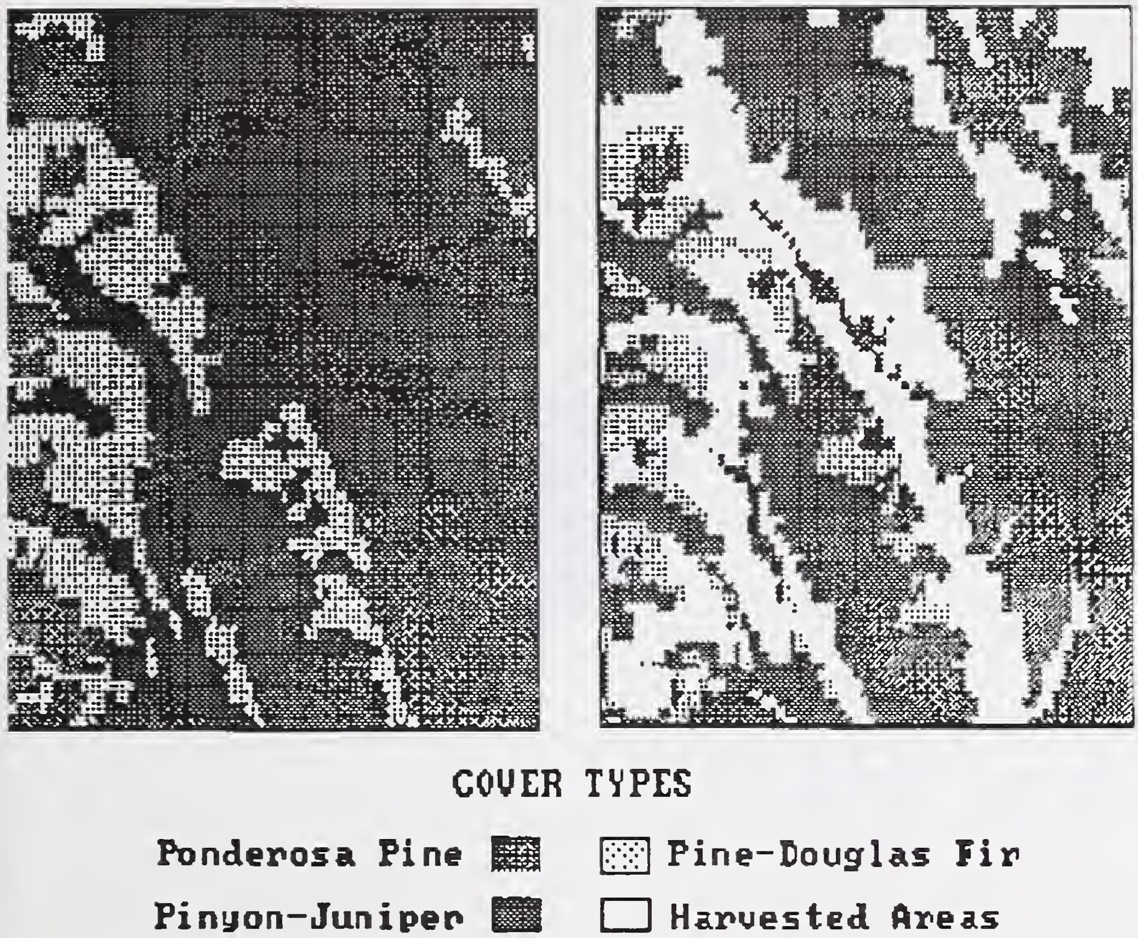

Figure 2.—Vegetation: left - no intervention, right - harvested.

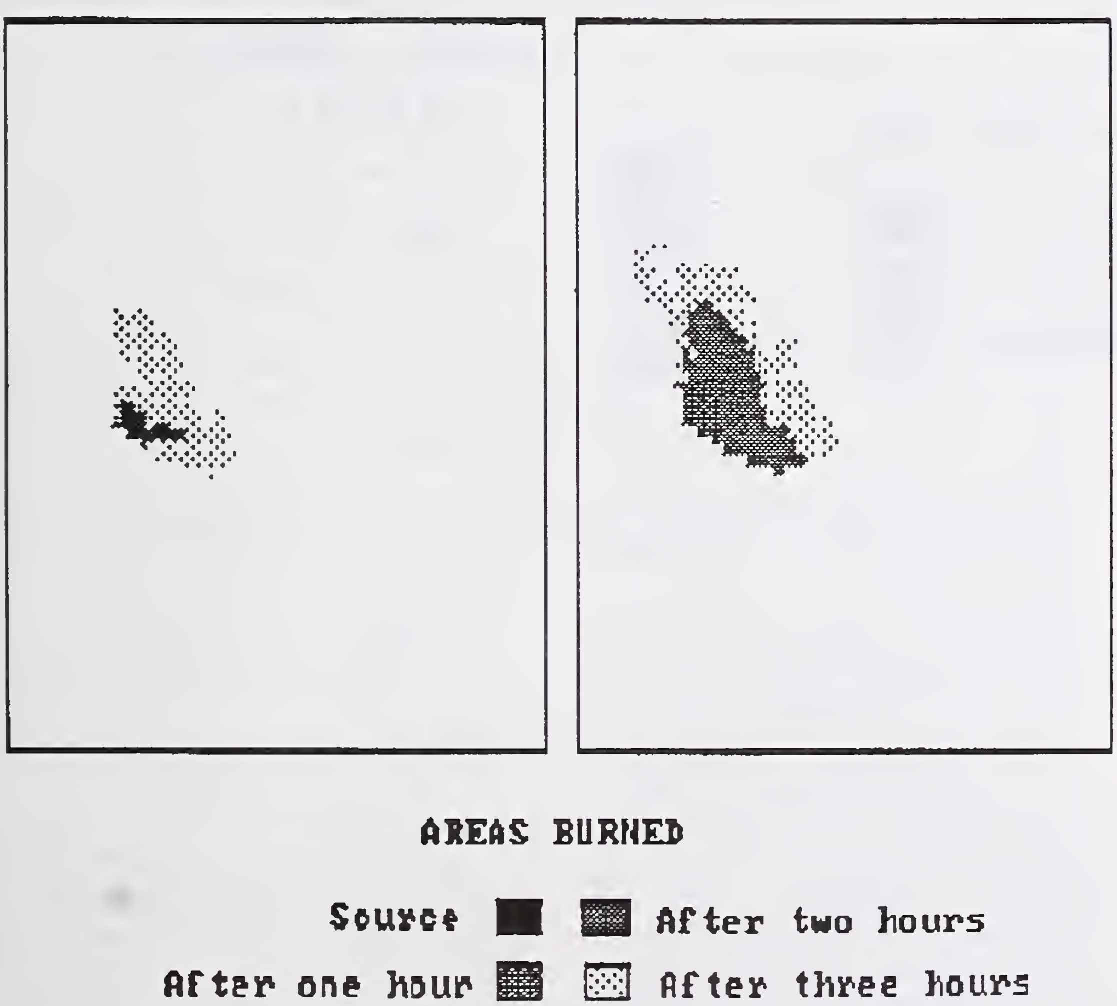

Figure 3.—Simulation results.

of Rothermel's fire spread model, and the predictions it makes are subject to the limitations and assumptions of the same model. However there are significant differences between this system and the BEHAVE system.

The capabilities of FIREMAP for spatial/temporal simulation of fire behavior make it a useful tool that goes beyond the simple display of spatially summarized, rapidly available information. FIREMAP simulates the actual spread of fire predicting its varying intensity and showing areas burned for chosen time intervals. It can be used for "on site fighting," thanks to MAP capabilities for quick information update (like a change in fuel types due to clear cutting) predicting the extent and intensity of a fire for a certain period of time.

In order to choose a fire management program it is necessary to consider not only ranges of fire behavior under various management alternatives, but also to assess the relative probability of occurrence of certain fire events. FIREMAP does not consider this latter aspect, a point that should not be overlooked in the decision-making process. It might be more practical to direct more attention on planning for those less probable ignitions that are likely to escape and cause extensive damage when they do occur (Salazar and Bradshaw 1986).

Future work with FIREMAP should include validation and sensitivity analysis, and fine tuning of the module presently running. The prediction capabilities can be greatly increased by addition of other modules, either from BEHAVE or new ones. For example, a module to simulate spotting, or a module to compute scorch height can be easily integrated.

The use of a more sophisticated GIS with flexible command language and built-in modelling modules, real number processing capabilities, and larger memory, will also help in

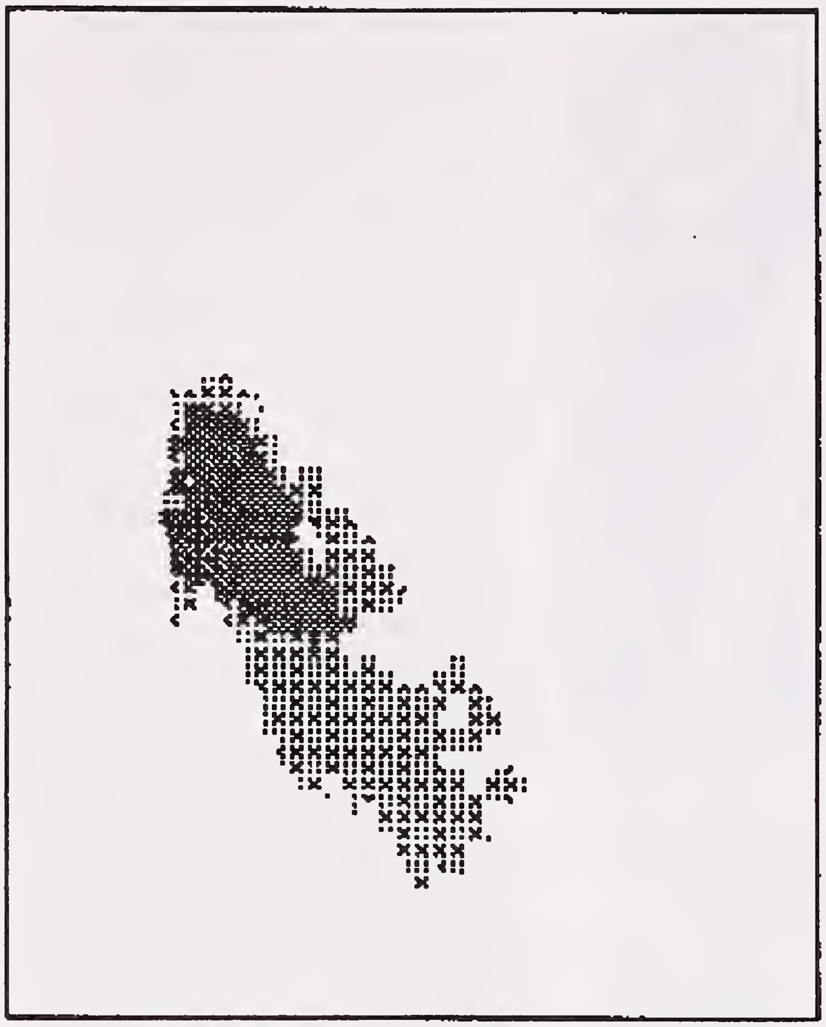

Figure 4.—Area burned with a wind change.

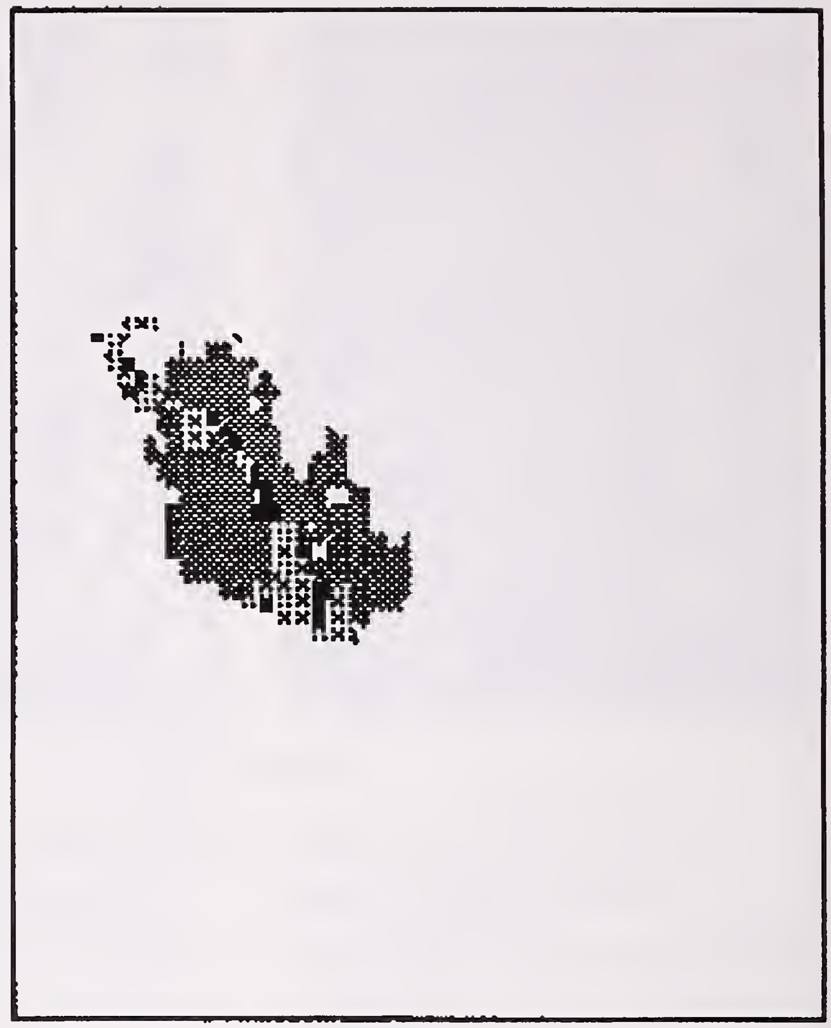

Figure 5.—Expected flame lengths.

overcoming some of the present limitations of FIREMAP.

Aknowledgements

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Streamflow and Water Quality Responses to Preharvest Prescribed Burning in an Undisturbed Ponderosa Pine Watershed[1]

Gerald J. Gottfried and Leonard F. DeBano[2]

Abstract.—Mean annual streamflow for the 6 years following burning did not increase significantly over pretreatment levels. Water quality changes were evaluated by comparing prefire and postfire levels of nitrate-nitrogen, ammonium-nitrogen, phosphates, calcium, magnesium, sodium, and potassium. Fire significantly changed the concentrations of some nutrients in stream water, but the changes were too small to adversely affect water quality.

Forest history studies (Arno 1980, Dieterich 1983) indicate that before fire suppression was initiated at the start of this century, most forest fires were surface fires. These fires reduced fire hazards and improved stand conditions by preparing seedbeds, thinning advance regeneration, and retarding the invasion of more shade-tolerant species. Current USDA Forest Service policy allows managers to use planned and unplanned fires for maintaining or enhancing resources (Arno 1980).

The effects of prescribed burning on streamflow and water quality have not been studied, although the effects of stand-replacing wildfires are well documented (Campbell et al. 1977, Tiedemann et al. 1979). The possible effects of prescribed burning for augmenting streamflow are of interest to forest managers. Dieterich (1983) hypothesized that prescribed burning, which reduces stand density and total forest floor depth, could increase runoff, or at least make more soil water available on a site. Watershed experiments (Rich 1972, Rich and Gottfried 1976) have shown increased runoff after the creation of openings or after severe reductions in stand density. The degree of density reduction would be critical if enhanced streamflow were expected. Haase (1986) measured greater surface soil water and subsequent ponderosa pine seed germination following burning.

Soil changes that increase the surface runoff would be expected to increase streamflow volumes and peak flows. Although intense fires can decrease infiltration into the soil by reducing porosity or creating hydrophobic conditions (DeBano 1981), prescribed burning has not produced surface runoff, or accelerated erosion, as long as the forest floor was not completely consumed during a fire (Biswell and Schultz 1957). Cooper (1961) studied the effects of controlled burning east of White River, Arizona, and found that although erosion and soil exposure increased, only small amounts of soil were moved, most of which never reached perennial streams. He also found the water-holding capacity of burned and unburned humus was similar and concluded prescribed burning did not influence streamflow and only slightly affected watershed condition.

Clary and Ffolliott (1969), working in an Arizona ponderosa pine stand, determined that the humus layer, the lowest layer of the forest floor, had to be modified, or removed, before the forest floor's ability to intercept and hold precipitation was significantly reduced. Agee (1973) found that prescribed burning reduced forest floor water-holding capacity in two California mixed conifer habitat types but concluded erosion could be avoided if the litter cover was not destroyed.

Several studies have shown available nitrogen increases following prescribed burning (Covington and Sackett 1986, Ryan and Covington 1986, Vlamis et al. 1955). A two-stage increase in nitrogen availability occurs. Immediately after burning, ammonium-nitrogen (NH_4-N) is high because of the pyrolysis of organic matter. High levels of NH_4-N are followed by increasing levels of nitrate-nitrogen (NO_3-N) when nitrification begins. Readily available phosphorus is also increased by fire (DeBano and Klopatek 1988, Vlamis et al. 1955).

Sims et al. (1981) measured chemical properties of water from small runoff plots before and after a controlled burn in a ponderosa pine-mixed conifer stand near Tucson, Arizona, and found that mean concentrations of calcium, magnesium, and fluoride increased significantly. However, it has not been determined whether an increase in readily available nutrients on a burned site increases nutrient loading in adjacent streams. This is important because higher concentrations of nutrients could adversely affect water quality for domestic and livestock consumption and lead to eutrophic conditions in affected aquatic ecosystems.

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Research Forester and Supervisory Soil Scientist, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Flagstaff and Tempe, AZ, respectively. Headquarters are in Fort Collins, in cooperation with Colorado State University.*

In 1981, a prescribed fire was ignited on the East Fork of Castle Creek, a gaged ponderosa pine-mixed conifer watershed by the Alpine Ranger District personnel. The main objectives were to reduce fuels with a minimum of mechanical disturbance, maintain fuel loading at a manageable level, and evaluate the effects of prescribed fire in virgin ponderosa pine and mixed conifer stands on the timber, watershed and forage resources. This prescribed burn provided us the opportunity to evaluate the impacts of fire on water augmentation and quality.

Table 1.—Average measurements of the West Fork and East Fork watersheds.[1]

Measurement	West Fork	East Fork
Size (acres)	900	1,163
Aspect (direction, average of plots)	S 43°E	N 14°W
Slope (average of all measured points, percent)	12.6	13.8
Forage production (pounds per acre)	78.2	119.8
Litter (pounds per acre)	33,177	31,085
Soil mantle depth (feet)	2.6	2.8
Forest conditions 1964 (per acre) before harvesting		
Basal area	135	122
Board-foot volume[2]	11,060	10,680
Forest conditions (per acre) before burning	1975	1981
Trees	328	608
Basal area	60	139
Board-foot volume[2]	2,759	11,843

[1]*Data prior to 1975 from Rich, Lowell R. 1972. Managing a ponderosa pine forest to increase water yield. Water Resources Research 8: 422-428.*

[2]*Scribner Rule.*

The Study Area

The East Fork and adjacent West Fork of Castle Creek (fig. 1) are within the Apache-Sitgreaves National Forests of east-central Arizona, approximately 14 miles south of Alpine. Elevations vary from 7835 to 8583 feet. Soils are derived from basalt. The primary soil subgroups on East Fork are Mollic Eutroboralfs, Lithic Argiborolls, and Eutric Glossoboralfs (Laing et al. 1988). Additional information on these watersheds is presented in table 1, primarily from Rich (1972).

Annual precipitation between 1956 and 1987 averaged 27 inches; the highest, 39.02, occurred in 1979 and the lowest, 16.86 inches, in 1974. Winter precipitation between October 1 through May 31 averaged 15 inches, about 57% of the annual total. Much of the winter precipitation occurs as snow, although occasional late fall rainstorms have produced large amounts of precipitation and accompanying peak stormflows. Summer precipitation is normally produced by convection storms during the regional monsoon.

Vegetation on Castle Creek has been classified as a *Pinus ponderosa* var. *scopulorum/Q. gambelii* type (Laing et al. 1988). Ponderosa pine accounts for 81% of the total basal area (table 1). Mixed conifer stands found on north-facing slopes and along drainages include ponderosa pine, Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), southwestern white pine (*P. strobiformis*), and quaking aspen (*Populus tremuloides*). Mixed conifer vegetation occupies about 303 acres on East Fork.[3] Gambel oak (*Quercus gambelii*) and

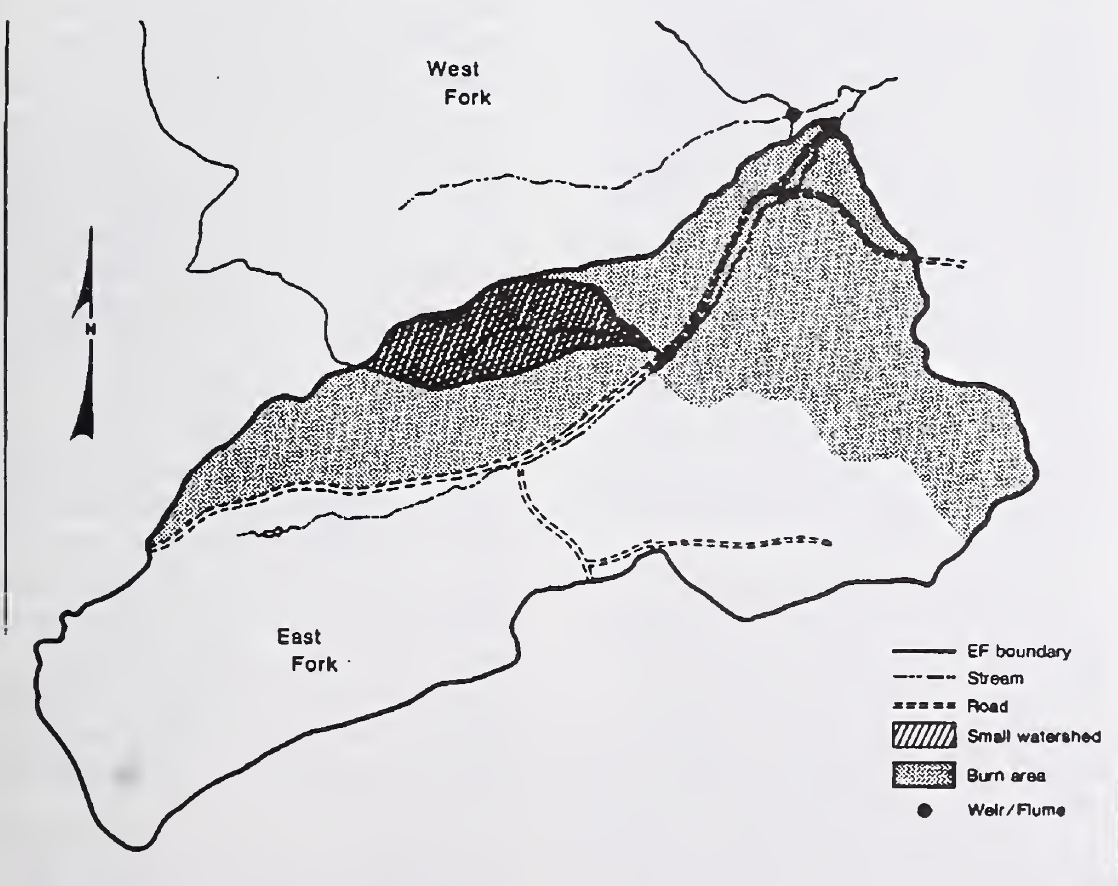

Figure 1.—The prescribed burn covered 43% of the East Fork of Castle Creek.

[3]*Soto, Edward L. 1981. Environmental analysis report: East Castle Creek Prescribed Burn, 23 p. USDA Forest Service, Apache-Sitgreaves National Forests, Alpine Ranger District, Alpine, AZ (Unpublished report).*

New Mexico locust (*Robinia neomexicana*) are found throughout both watersheds. The stream channel runs through a 68-acre meadow in the upper part of Castle Creek East.

Watershed History

The two experimental watersheds were established in 1955 when 120° v-notch weirs were constructed across the intermittent streams on each drainage. The original objective on Castle Creek was to evaluate the effects of an improved type of timber harvesting on water and sediment yields, wildlife and scenic values, and on the timber resource (Rich 1972). Both watersheds were calibrated for 10 years (1956-1965); mean annual flow during this period was 1.97 ± 0.61 inches on West Fork and 2.97 ± 0.89 inches on East Fork. In 1966, West Fork was harvested so that one-sixth of the watershed was clearcut in blocks fitted to stand conditions, and the remaining area was put into the best growing condition by removing poor risk, overmature, and diseased trees and releasing residual trees (Rich 1972). East Fork served as the hydrological control. The blocks were planted with ponderosa pine seedlings. Rich (1972) reported this treatment increased the average water yield by 29%; any streamflow increases greater than 0.4 inch were statistically significant.

The Prescribed Fire

Before the burn, Sackett (1979) measured 31.9 tons per acre of fuel on the East Fork; 47% of the fuel was in rotten material over 3 inches in diameter. Total dead fuel load (of 31.9 tons per acre) on Castle Creek was greater than the average 21.7 tons per acre measured in relatively undisturbed stands throughout Arizona and New Mexico.

The burning plan specified a 70% reduction in fine fuels and a 40% reduction in heavy fuels. Burning began during the first week of November 1981; however, an administrative decision was made to terminate the prescribed burn before completion because the burn did not appear to be meeting fuel reduction objectives. Consequently, only 503 acres, 43% of the watershed, was actually burned (fig. 1). The burned blocks were on the south-facing slopes, and on north-facing slopes in downstream areas near the weir. Ninety-three percent of the trees in these areas were ponderosa pine. An average acre in the burned zone contained 624 trees, 132 square feet of basal area and 9,610 board feet. The burned area contained 73 permanent timber inventory points.

A formal evaluation of fuel consumption was never conducted. However, Michael Harrington, research forester with the Station, reporting on his observations on November 4 and 5, 1981, considered fuel consumption satisfactory from a fire hazard standpoint considering the dense, deep forest floor common in undisturbed stands. Surface fuels were consumed and the middle forest floor layers were only slightly charred over most of the southern exposures. The duff layer was only consumed near sawtimber trees or adjacent to heavy fuels. Few of the downed logs were totally consumed. Mixed conifer pockets burned poorly. Although more fuel consumption would have occurred if fuel and weather conditions had been warmer and drier, the dense duff layer would probably not have been completely consumed even under ideal conditions.

Changes in the residual stand were minimal: only 4% of the trees sampled within the burned blocks using point sampling techniques showed evidence of more than 10% crown scorch. This was equivalent to 79 trees per acre or 13% of the total stand; 89% of the damage was in the 2- and 4-inch diameter (d.b.h.) classes. Approximately 78% of the scorched trees contained over 90% scorch, 42% of these trees subsequently died by 1986. However, most of the damage and mortality was confined to 3 of the 73 points; 11% of the stocked points were completely unburned. Mortality was equivalent to 1% of the preburn average basal area, and to 0.2% of the board-foot volume. No new basal scars were found on the sample trees.

Methods

Streamflow

We used a paired watershed approach to analyze the impacts of prescribed burning on water yields. Prefire regressions were developed between East Fork and West Fork, the control watershed (fig. 2). Similar regressions were prepared after the treatment period (1982-1987),[4] and the two were compared by covariance analysis to determine whether significant changes occurred. The October 1 - September 30 water year was used for all hydrological analyses. The current analyses was unusual because we reversed control watersheds for this experiment; East Fork had been the control for an earlier West Fork harvesting experiment (Rich 1972). This reversal only works if the relationship between the two watersheds has remained constant since the initial harvest treatment. To check for this we developed a linear regression for the 15-year postharvest period (1967-1981). The regression (fig. 2) had a coefficient of determination (r^2) of 0.994 and a standard error of 0.323 indicating the relationship had remained constant. The standard error for the postfire regression was 1.072. Since many watershed studies have shown a decline in treatment effects with time, the prefire relationship was also checked for

[4] *Colmer, Gerald K. 1988. Castle Creek hydrologic data for the 1984-1987 water years. USDA Forest Service, Apache-Sitgreaves National Forests, Springerville, AZ.*

changes over time using a technique described by Baker (1986), but again no significant influence was noted. Long-term mean annual runoff for West Fork (1967-1987) was used as our average independent variable (x) for calculating percent changes in water yield. This made the analysis less sensitive to extremely high or low streamflows, and gave a better indication of average changes. The same regression techniques and appropriate long-term means were used for analyzing seasonal and monthly runoff changes. Statistical significance was indicated by values above the 5% level.

Water Quality

Stream water samples were collected at the main gaging station of Castle Creek East (CC-East) and at a small flume installed at the base of a subwatershed (CC-Sub) in CC-East (fig. 1) during the snowmelt periods immediately preceding (spring 1981) and following the prescribed fire (spring 1982). A small Parshall flume had been installed in October 1979 to measure streamflow originating from the 61-acre subwatershed on the East Fork of Castle Creek. Water samples were collected at both gaging stations twice daily at 0500 and 1700 hours, when minimum and maximum stream discharges commonly occur during snowmelt in eastern Arizona. Sample bottles were charged with phenyl mercuric acetate (PMA) to eliminate microbial activity. Ambient temperatures were near freezing for most of the collection period, which further minimized changes in the water chemistry.

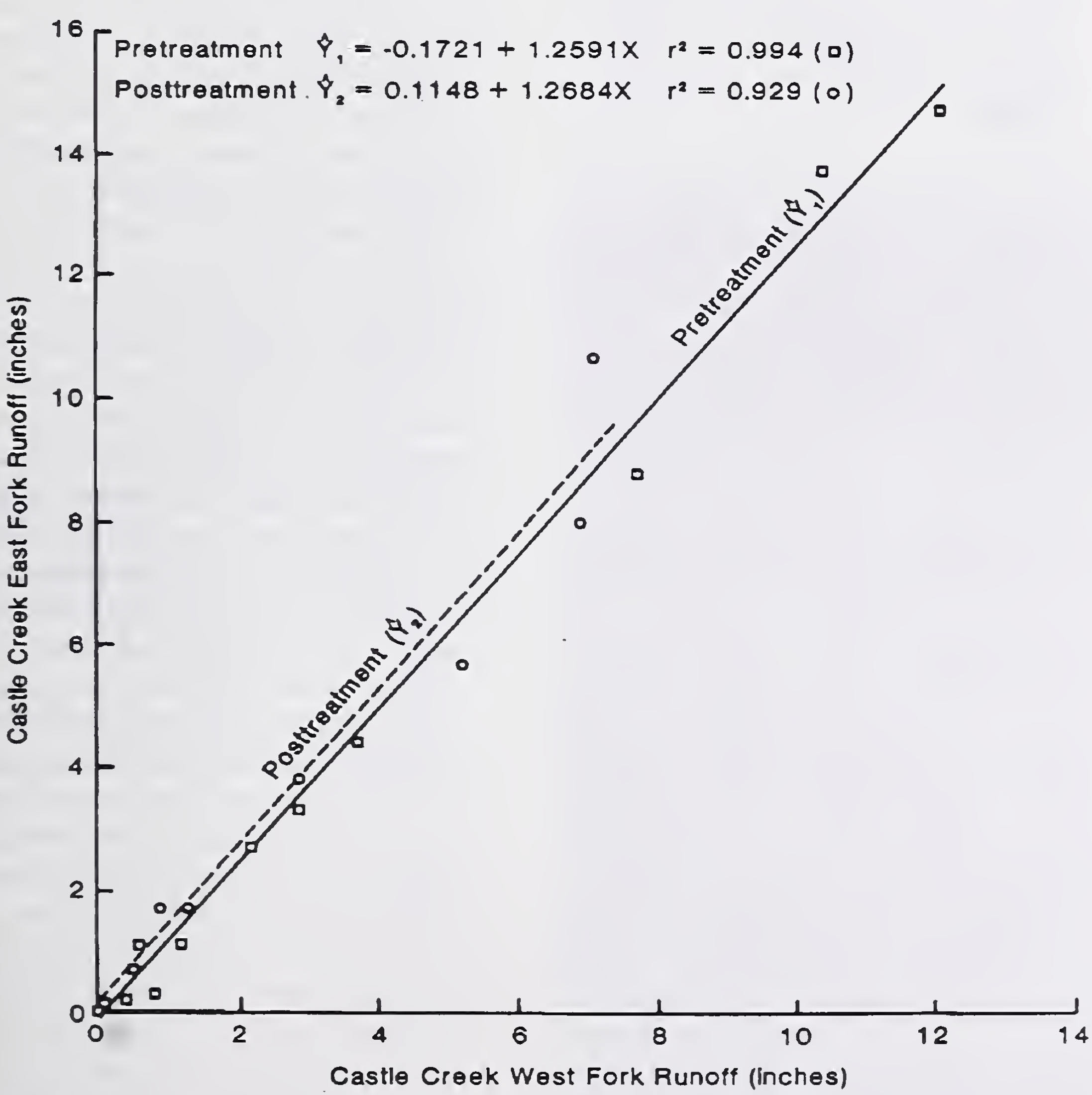

Figure 2.—The prescribed fire did not produce significant changes in annual streamflow; the pretreatment and posttreatment regressions were similar.

Water samples were analyzed for NH_4-N, NO_3-N, orthophosphate, calcium (Ca), magnesium (Mg), sodium (Na), and potassium (K). Concentrations of N and P compounds were determined colorimetrically, and concentrations of the cations were determined using atomic absorption. All concentrations were reported in parts per million.

Changes in water quality were tested for statistical differences by using t-tests on averages of nutrient concentrations obtained during three snowmelt periods between February and May in 1981 (prefire) and 1982 (postfire). The three periods during snowmelt were: the first 10 days, the second 10 days, and the remainder of snowmelt. Although arbitrary, these periods were selected because we expected soluble nutrients produced by burning to dissolve and leave the watershed during initial snowmelt. Separate analyses were used for each watershed and nutrient for each of the three snowmelt periods.

Results

Streamflow

The prescribed fire on East Fork, did not significantly increase average annual water yield (fig. 2). Analysis of six years (1982-1987) data during posttreatment showed streamflow increased 0.32 ± 0.70 inch (8% ± 18%) for the entire watershed, but this amount was small enough to have occurred by chance. If this increase were prorated over the burned area only, it would be equivalent to 0.74 inch, or an increase of about 19%. Analyses of pretreatment and posttreatment regressions for the 8-month winter period and for the summer period also showed fire did not significantly increase seasonal streamflow.

Evaluations of monthly runoff volumes indicated no statistically significant changes for January; for March and April, the two months

with the greatest runoff; and for July and August, the two driest months. Analyses for the other seven months indicated some generally small but statistically significant differences. The February data showed the largest monthly increase of 0.11 ± 0.08 inch or 46% ± 34%. Increases of 0.03 ± 0.04 inch (24% ± 36%) were indicated for May and 0.02 ± 0.01 (26% ± 17%) for September. The other four months showed declines in runoff, but again the amounts were very small. Some of these differences may reflect the smaller data range in the posttreatment period rather than changes in actual hydrological processes.

Water Quality

Nutrients in stream water produced during snowmelt on both the large watershed (CC-East) and the subwatershed (CC-Sub) reflected the effects of prescribed burning (table 2). Concentrations of NH_4- and NO_3-N in stream water from CC-Sub increased upon the onset of spring runoff in contrast to CC-East where the increases occurred 10-20 days following the beginning of snowmelt. Phosphorus concentrations in stream water did not change significantly as a result of burning with the exception of a small, but significant, increase of PO_4 in the stream water leaving CC-Sub during the first snowmelt period (table 2).

The responses of cations to burning was variable (table 2). Changes in concentrations of Ca and Mg in response to burning were inconsistent between the two watersheds. Concentrations of K in stream water from CC-Sub increased significantly during the first two snowmelt periods following burning, in contrast to CC-East where it increased only during the second snowmelt period. Little change in concentrations of Na occurred in stream water as a result of burning.

Table 2.—Nutrient concentrations (ppm) of streamflow from Castle Creek East Fork (CC-E) and a small subwatershed (CC-Sub) during snowmelt periods before (spring 1981) and after (spring 1982) a moderately intense prescribed burn in a ponderosa pine forest in eastern Arizona.

Watershed		Nutrient						
Snowmelt period[1]	Fire	NH_4-N	NO_3-N	PO_4	Ca	Mg	K	Na
CC-Sub								
0-10	Pre	0.0017a[2]	0.0002a	0.32a	10.08a	6.95a	0.20a	2.48a
	Post	0.1550a	0.0018b	0.37a	10.06a	6.82a	0.38b	2.59a
11-20	Pre	0.0020a	0.0006a	0.32a	10.26a	6.90a	0.18a	2.58a
	Post	0.0222b	0.0016b	0.34a	9.62	6.48b	0.32b	2.58a
21+	Pre	0.0104a	0.0013a	0.33a	12.20a	8.12a	0.27a	2.89a
	Post	0.0101a	0.0029b	0.34a	10.68b	7.34b	0.31a	2.71a
CC-East								
0-10	Pre	0.0196a	0.0020a	0.47a	9.76a	5.93a	0.97a	2.42a
	Post	0.0251a	0.0019a	0.53b	11.45b	6.63b	1.05a	2.65a
11-20	Pre	0.0039a	0.0010a	0.49a	11.22a	6.71a	0.85a	2.76a
	Post	0.0104b	0.0010a	0.49a	12.77b	7.60b	0.98b	2.82a
21+	Pre	0.0041a	0.0004a	0.49a	15.72a	5.37a	0.77a	2.84a
	Post	0.0124b	0.0019b	0.49a	12.93b	8.00b	0.74a	2.88b

[1]*Days since the start of snowmelt.*

[2]*Prefire and postfire values for individual watersheds and nutrients having the same letter are not significant at the 0.05 level.*

Discussion

The prescribed fire did not significantly increase streamflow volumes. This is not surprising because forest conditions were not really affected by the fire, only 5% of the trees per acre and 1% of the preburn basal area per acre were destroyed. Most water yield increases in southwestern forests occurred when mature conifers were replaced by grass, herbaceous species, or by conifer seedlings (Rich 1972, Rich and Gottfried 1976). The replacement vegetation uses less soil water during the growing season, and consequently less winter precipitation is needed to recharge the soil. Also, there is an earlier and more efficient movement of soil water into the stream system. On East Fork, no large openings were created—even the heavily damaged areas remained stocked. Some streamflow increases have also been attributed to partial cutting. The single-tree selection harvest at Workman Creek produced a small (0.23 inch) but statistically significant runoff increase (Rich and Gottfried 1976); Troendle and King (1987) have reported increases from partial cutting during wet years in Colorado. However, the stand loss on East Fork was much less than that which would have occurred in even the most conservative timber harvesting operation. A more severe reduction in stand density could have produced water yield increases, but it is uncertain how large a reduction would have been necessary.

Fire consumed little of the forest floor, which protects the soil and enhances infiltration. Most water movement on Castle Creek occurs as subsurface flow. A more complete loss of the forest floor could have produced more soil surface runoff and increased streamflow but at the cost of increased soil erosion.

The small but statistically significant changes in monthly streamflow were interesting even if they did not affect annual or seasonal volumes.

The largest increase, in February, could be related to higher infiltration during initial melting periods because of forest floor depth reductions and to more rapid snowmelting because of charred slash and tree trunks, which absorb more heat. Early season snowmelt, after soil recharge, is more efficient because evapotranspiration is low, and more water can reach the channels. Unfortunately, postfire measurements of forest floor depth, soil water, and infiltration to the soil were not made. May and September increases could be related to more water reaching the soil and less being held by, or evaporating from, the forest floor. Greater soil water on relatively deep soils results in longer runoff periods in the spring and greater runoff during the late summer. The decline in June flows could be related to higher flows in February and May, but this study could not demonstrate this. However, these volumes are too small to be considered in management planning.

It is unfortunate that the entire project area could not have been burned. However, the fact that only 43% of the watershed was treated does not preclude the possibility of measuring water yield changes. Increases were detected on the North Fork of Workman Creek when 32% of the watershed was treated (Rich and Gottfried 1976) and on the West Fork of Castle Creek when only 16% of the area was cleared (Rich 1972). The fact that burned areas were concentrated adjacent to the weir or channel (fig. 1) would have enhanced the chances of detecting increases because transmission distances, and resulting losses, would be smaller. We cannot assume that a more complete burn would have resulted in significant changes; variations in stand density, soil depth, water storage capacity, topography, or distances to stream channel lead to variations in a site's potential for water yield improvement even under severe vegetation reduction. East Fork is, in fact, a more realistic example of what could occur on a larger watershed where it is unlikely that the entire area would be burned.

Most changes in stream water chemistry in response to prescribed burning were expected. The increased concentrations of NH_4- and NO_3-N in stream water leaving the two gaged watersheds following fire probably occurred because NH_3 produced during the fire volatilized slowly over winter, became trapped in the snow pack, and remained there until spring snowmelt. Ammonia-N can remain high in soil for several months following fire before being decreased by nitrification and further volatilization. The increased concentrations of NO_3-N in the stream water following burning probably reflect nitrification because NO_3-N is usually not formed directly by burning. Phosphorus did not respond as expected because fire usually releases significant quantities of highly available phosphorus (DeBano and Klopatek 1988). The inconsistent increases and decreases in Ca and Mg between CC-Sub and CC-East cannot be explained. The increases in concentrations of K in the stream water leaving CC-Sub following burning probably represented losses of highly mobile K produced during burning.

Conclusions

The prescribed fire, which covered 43% of a previously undisturbed ponderosa pine watershed, did not result in statistically significant increases in annual or seasonal streamflow. These results were anticipated because damage to the forest stand was minimal, and the forest floor remained intact even though surface fuels were generally consumed.

Although statistically significant changes in nutrient concentrations occurred as a result of the prescribed fire on Castle Creek East, the changes were very small and of little consequence in terms of site productivity or downstream water quality. Changes in N and P compounds, which are important from the standpoint of water quality, only changed a fraction of one part per million, which is insignificant in terms of adversely affecting water quality.

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Survival of Damaged Singleleaf Pinyon One Year After Wildfire[1]

David R. Weise[2]

Abstract.—At two wildfire sites in southern California, first year postfire survival of singleleaf pinyon *(Pinus monophylla)* was 96% and 28% for trees with less than 67% crown scorch. Preliminary findings indicate that dbh and height may not affect survival of singleleaf pinyon after wildfire.

Pinyon-juniper woodland occupies 2.7 million acres in California (Bolsinger 1980). Singleleaf pinyon (*Pinus monophylla* Torr. & Frem.) is a common species of this type. Approximately 160,000 acres (or 20%) of the San Bernardino National Forest (SBNF) in southern California is covered by singleleaf pinyon. The type is heavily used for recreation on the SBNF and needs to be maintained.

Most of the large fires in singleleaf pinyon on the SBNF have been human-caused (Doherty, pers. comm.). Prescribed fire may be a method to maintain singleleaf pinyon stands by reducing fire hazard or preparing seedbeds (by reducing competition). This use is a departure from traditional use of fire in this type which has been eradicating the pinyon-juniper type throughout most of its range to improve forage production for livestock (Arnold et al. 1964, Wright et al. 1979).

The major factor governing the use of prescribed fire in singleleaf pinyon is the species' ability to withstand the damaging effects of fire. No information concerning survival of singleleaf pinyon after fire was found in the literature. One study (Dwyer and Pieper 1967) reported 13.5 percent mortality of common pinyon (*Pinus edulis* Engelm.) 1 year after a damaging wildfire. A pilot study to document postfire survival of singleleaf pinyon was established on the SBNF in 1987. This paper reports first year results of the study.

Study Sites

Two wildfires occurred in pure singleleaf pinyon stands on the SBNF in 1987. One fire (Nelson), located at 7000 ft elevation, burned approximately 40 acres in early July. The other (Jeep Trail) is located at 4800 ft elevation and burned 1600 acres in September. Both sites receive precipitation in the form of snow and occasional summer thunderstorms. The fuels adjacent to both wildfire sites are similar—light, patchy grasses and small, scattered brush. The Jeep Trail site has some heavier brush such as manzanita in scattered areas, primarily on lower slopes along intermittent drainages.

Methods

A total of 244 trees were tagged and measured at the study sites in October 1987. Trees were assumed to be living at the time of the wildfires. Diameter at breast height (dbh), total height, and crown damage class were assessed for each tree (table 1). Time constraints did not permit equal replication of all damage categories. Control trees (damage category 1) were located in unburned areas within 50 ft of the wildfires' boundaries. The sites were revisited in September 1988 and survival of each tree was assessed. Trees were placed into 1-inch dbh classes. Survival by dbh class and damage category was summarized by individual site.

Results

Tree size differed considerably between the two sites: trees selected at Nelson were smaller (table 2). The two sites differed greatly in first year postfire survival (table 3). A greater percentage of trees were still alive at the Nelson site. Survival does not appear to be a function of only dbh class (table 3)—the 24 inch tree that died at Jeep Trail was a damage category 2 tree.

Similarly, postfire survival differed by damage category at each location (table 4). All trees in categories 1 and 2, and 96% of trees in categories 1, 2, and 3 survived at Nelson. In contrast, only 36% of the trees in categories 1 and 2, and 28% of trees in categories 1, 2, and 3 survived at Jeep Trail. Twenty-five percent of the category 1 trees at Jeep Trail died during the first year. Survival response was similar at both sites in damage categories 4-7. Only two of the 158 trees in these categories survived.

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Forester, Forest Fire Laboratory, Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture, Riverside, CA.*

Discussion

The differences in first year postfire survival are of interest. Mortality of the undamaged and slightly damaged trees appears to differ between the sites. Since fire behavior on either fire is not adequately documented, determination of the factors causing the difference may be difficult. This is often the case when trying to link wildfire behavior to fire effects.

Tree size does not appear to be a factor in initial survival. However, tree size may prove to be critical in long term survival. Mortality of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) nearly doubled between 10 and 22 months after a prescribed fire (Wyant et al. 1986). Postfire mortality of singleleaf pinyon may change considerably over the second year also.

Attributing mortality to only fire damage may not be possible. Since 25% of the undamaged trees at Jeep Trail died during the first year, other factors may be contributing to mortality. The species is known for cold hardiness (Hepting 1971), but the interaction of fire damage with low winter temperatures may have contributed to mortality. Singleleaf pinyon may achieve longevity of 225 years and diameters of 12 inches are unusual (Hepting 1971). The Jeep Trail trees, with mean diameter of approximately 12 inches, may have reached maturity and trees have begun to die naturally. The fungus *Verticicladiella wagenerii* is also known to occur on the SBNF. The factors of age, cold, and root rot may have contributed to mortality of the undamaged trees.

A relationship between percent of crown scorch and singleleaf pinyon survival may exist. The species appears to be able to tolerate a small portion of crown scorch resulting from fire (less than 33 percent). It may be possible to burn a stand with a low intensity prescribed fire and keep crown damage to a minimum. Additional measurements planned

Table 1.—Description of crown damage categories.

Damage category	Description
1	Control tree, no damage
2	Less than 33% crown scorch
3	33% to 66% crown scorch
4	More than 66% crown scorch
5	Complete scorch, less than 33% crown consumption—needles and small branches burned
6	33% to 66% crown consumption
7	More than 66% consumption

Table 2.—Stand summary for singleleaf pinyon study sites.

Damage Category	Stems Nelson	Stems Jeep	Mean dbh Nelson	Mean dbh Jeep	Mean height Nelson	Mean height Jeep
			(inches)		(feet)	
1	10	8	3.7	9.7	12.8	23.5
2	21	17	4.2	10.6	13.4	24.3
3	14	14	4.1	6.5	13.4	22.1
4	19	20	3.9	8.0	12.4	20.1
5	17	21	2.8	7.0	11.5	18.5
6	21	21	4.5	8.0	14.1	20.2
7	21	20	4.5	8.0	14.7	23.8

Table 3.—Summary (number of trees) of singleleaf pinyon first year post-fire survival by diameter class and location.

Diameter class	Nelson Live	Nelson Dead	Jeep Trail Live	Jeep Trail Dead
0	4	4	0	0
1	4	6	0	4
2	8	11	0	7
3	5	20	1	4
4	8	12	3	11
5	4	6	1	7
6	4	4	1	9
7	2	6	1	12
8	0	2	1	7
9	2	4	0	6
10	1	2	0	11
11	2	1	0	8
12	0	0	0	3
13	0	1	2	7
14	0	0	0	6
15	0	0	0	2
16	0	0	0	0
17	0	0	1	2
18	0	0	0	1
19	0	0	1	0
20	0	0	0	1
21	0	0	0	0
22	0	0	0	0
23	0	0	0	0
24	0	0	0	1
Total	44	79	12	109

Table 4.— Summary (number of trees) of singleleaf pinyon first year post-fire survival by damage class and location.

Damage category	Nelson Live	Nelson Dead	Jeep Trail Live	Jeep Trail Dead
1	10	0	6	2
2	21	0	3	14
3	12	2	2	12
4	1	18	1	19
5	0	17	0	21
6	0	21	0	21
7	0	21	0	20
Total	44	79	12	109

within the next year should clarify this relationship.

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245

Fire and Forest Insect Pests[1]

J. M. Schmid and D. L. Parker[2]

Abstract.—Fire is used to kill bark beetle infestations in logging residuals or infested trees but is rarely used to suppress extensive infestations. Although fire may suppress populations of insects that spend a portion of their life cycle inhabiting the litter, conditions are not usually conducive for satisfactory suppression or are often too hazardous for burning. Forest fires predispose stands to attack by wood boring beetles and bark beetles as well as creating stand conditions conducive to extensive infestations in the future.

Forest fires affect insect populations by killing individuals and modifying the environment in which they live. By the same token, insect populations affect the forests and frequently increase the fire hazard. For purposes of this paper, insects will refer to only those species usually considered forest pests. This consideration ignores the multitude of forest insects that do not directly affect human activities. Such species are, therefore not subject to various suppression methods although they may be affected by them.

Fire is used to kill populations of insects inhabiting the bark or wood portion of trees but is rarely used to kill foliage insects. The branches, and cull logs created by logging activities, provide suitable habitat for some species of bark beetles—the most detrimental being in the genera *Ips* and *Dendroctonus*. The infestation of logging residuals in itself is not detrimental, but the populations developing within the logging residuals frequently infest living trees upon emergence if no additional residuals are available. In pine, *Ips*-infested slash is piled and burned to kill the brood. In spruce, logging residuals—mostly cull logs—are also piled and burned to suppress spruce beetle populations. Because the bark beetle brood develops within the phloem of the bark, the entire woody portion of the branch or log does not have to be consumed, just seared or heated to the point of destroying the inner bark or killing the inhabiting brood. Burning beetle-inhabited logging residuals is useful in suppressing endemic populations but is impractical in suppressing populations during outbreaks.

Fire is also used to kill populations of insects in standing trees or as part of felling and burning techniques. Standing *Dendroctonus*-infested pine are cut, piled, and burned when small infested groups occur. This technique is used to suppress small, scattered infestations of beetles but is rarely used during extensive outbreaks.

Prescribed burning has seldom been used to kill insects, primarily because few species inhabit the ground surface during any part of their life cycle. Lepidopterous species (moths) are found in the litter more than any of the other major insect pests because various species pupate in the litter or upper few centimeters of soil. Prescribed burning has been used against one species—the pandora moth—with limited success. Early summer burns killed about 60% of the pupae. Mortality was limited by the sparse, interrupted distribution of the litter, which prevented the fire from killing pupae in the bare soil.

Early summer burns coincide with periods of high fire danger. Thus, their use may be restricted because of possible conflagrations arising from the prescribed burning. Fall burning is more satisfactory from the fire danger standpoint. However, the very conditions that reduce the fire danger—moister litter, cooler air temperatures—also inhibit the buildup of sufficient heat to cause pupal mortality. Fall burning is less successful than early summer burning.

If prescribed burning against the pandora moth is an indication of the success of the technique against other insects, then prescribed burning has limited use in suppressing ground-inhabiting forest insects. However, the success of prescribed burning may depend to a great extent on insect habits as well as litter accumulations. If the insect is like the pandora moth—pupating in essentially bare soil as well as in and beneath the litter—then a fire will cause less than acceptable mortality. If the insect pupates or overwinters solely in the litter—e.g., cone beetles—then fire may cause consistently higher mortality.

In contrast to insect suppression, prescribed burning may predispose trees to the attacks of bark beetles and wood-borers. In the Southwest, large ponderosa pines usually have a thick deposit of litter around the

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[2]Schmid is Research Entomologist, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO 80526; Parker is Entomologist, Forest Pest Management, Southwestern Region, Albuquerque, NM.

trunk at ground level and this generates high temperatures during prescribed burning. If these trees are not killed outright, they can be severely damaged and then become infested by the western pine beetle, *Dendroctonus brevicomis* LeConte, and die. Prescribed burning of mature pine on forests such as the Kaibab National Forest may also just char the bark at the bases of the trees. These trees may be subsequently infested by the red turpentine beetle, *Dendroctonus valens* LeConte, which infests the lower portion of the bole. While red turpentine beetle attacks rarely kill the tree, they do dispose it to other bark beetles that may kill it. Also attracted to fire-scorched trees are wood-borers—roundheaded (cerambycid) and flatheaded (buprestid) beetles. These borers are abundant in trees damaged or killed by wildfires. They cause more damage than the red turpentine beetle because they bore into the woody tissue and may create extensive galleries within the wood, thus degrading the lumber.

Forest fires as well as the lack of them have frequently been cited as promoting stand conditions conducive to extensive infestations. Forest fires in the late nineteenth century created extensive, high-density, even-aged, lodgepole pine stands. These stands have now reached the size to be highly susceptible to the mountain pine beetle. Extensive infestations of the mountain pine beetle are now occurring in the northern Rockies. Similarly, the lack of widespread fires in the mixed conifer forest type has been cited as the reason for the development of high-density, uneven-aged, multistoried stands. Such stands, when composed mostly of Douglas-fir or Douglas-fir and white fir, are highly susceptible to outbreaks of the western spruce budworm and Douglas-fir tussock moth.

Forest fires generally influence insect populations but sometimes insect populations influence forest fires. Bark beetle-killed trees increase the short-term fire hazards. The dry needles and fine branches of recently killed trees increase the fine fuel component of fire danger. This danger remains high until the needles and twigs are lost from the dead trees—3 to 5 years after tree death. The larger branches and boles of dead trees increase the large fuel component for fires. As a result, any fire will be more severe. This increase in fuel loading will last for 10-20 years for pine and 30-50 years for spruce. Eventually the trees fall and become part of the ground-fuel component until they deteriorate.

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245

Interactions of Fire and Dwarf Mistletoe on Mortality of Southwestern Ponderosa Pine[1]

Michael G. Harrington and Frank G. Hawksworth[2]

Abstract.—A ponderosa pine stand infected with dwarf mistletoe was prescribed burned in Grand Canyon National Park. The degree of dwarf mistletoe infection positively influenced the degree of crown scorch. Amount of scorch was the dominant factor in first-year mortality. However, in the medium scorch classes, tree mortality increased as infection increased. Results suggest that dwarf mistletoe can be managed with prescribed fire.

Little is known about the effect of fire injury on pines infected with dwarf mistletoe (*Arceuthobium* ssp.) (Alexander and Hawksworth 1975). A prescribed burn on permanent plots in ponderosa pine (*Pinus ponderosa* subsp. *scopulorum* [Engelm.] E. Murray) on the South Rim of Grand Canyon National Park, Arizona, enabled us to compare fire-kill in trees with various intensities of dwarf mistletoe infection (*Arceuthobium vaginatum* subsp. *cryptopodum* [Engelm.] Hawksw. and Wiens).

Although fire has long been considered the primary natural control agent of dwarf mistletoe (Hawksworth 1961, Roth 1953), little quantitative information on the subject is available, particularly for ponderosa pine. Koonce and Roth (1980, 1985) studied the effects of prescribed fire on survival of ponderosa pines affected by a similar dwarf mistletoe (*Arceuthobium campylopodum* Engelm.) in Oregon. They concluded that the dwarf mistletoe could be partially sanitized from thinned and unthinned ponderosa pine stands by prescribed understory burning. Their studies were conducted in immature, even-aged stands and not in uneven-aged stands such as we describe here for the Grand Canyon study. The objective of this study was to determine the effect of various combinations of crown scorch and dwarf mistletoe infection on first-year mortality of ponderosa pine.

Methods

The Grand Canyon plots were established in 1950 as part of a dwarf mistletoe control project (Lightle and Hawksworth 1973, Maffei 1984). A portion of the study area was treated for dwarf mistletoe by pruning or cutting infected trees. The rest of the study area was left untreated as a control. This paper presents results from the untreated study area only. From 1950 to 1982, stand characteristics and dwarf mistletoe infection had changed considerably. Stand density for trees greater than 3 inches diameter breast height (d.b.h.) decreased from 29 trees per acre to 19 trees per acre, while average d.b.h. increased from 14.5 to 15.6 inches. The percentage of infected trees has not changed from about 80%, but the average dwarf mistletoe rating increased from 2.8 to 3.7 on the 6-class scale. In this 6-class system, the tree crown is divided horizontally into thirds. Each third is given a rating of 0 (no mistletoe), 1 (light mistletoe), or 2 (heavy mistletoe). The three ratings are totaled to give a tree rating that may range from 0 (no mistletoe) to 6 (entire tree heavily infected) (Hawksworth 1977).

Fire Treatment

The dwarf mistletoe study area was prescribed burned by the National Park Service as part of a 600-acre fuel and stand density reduction burn. The burning began August 9, 1985, and continued until heavy rains extinguished the flames on August 24 (Ray 1985). After fire lines were burned out, fires were allowed to spread at will over much of the area. Fire intensities were generally low to moderate throughout the 15 days of burning, with the exception of some crowning in a 1-acre sapling thicket as heavy fuels burned with gusty winds. Most of the crown scorching resulted from burning of heavy surface fuels. Over the 600 acres, the fire reduced the average duff depth of 1 inch by about 40% and the total down woody fuel loading of 5.6 tons per acre by 33%. Ponderosa pine saplings less than 3 inches d.b.h. were reduced by 52% from a preburn average of 320 trees per acre (Ray 1985). Weather and fuel moisture information from the fire was not available.

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Harrington is a Research Forester, Intermountain Fire Sciences Laboratory, Intermountain Research Station, Missoula, MT 59807. Hawksworth is a Research Plant Pathologist, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO 80526.*

Damage Evaluation

This prescribed burn provided an opportunity to study the effects of fire damage on the 191 sample trees in the untreated plot that had been given dwarf mistletoe ratings (DMR) in 1982. Each tree was given one of five crown scorch ratings based on percentage of crown length that was scorched. Key scorch values and approximate ranges are as follows: 0% (0-12%), 25% (13-37%), 50% (38-62%), 75% (63-87%), and 100% (88-100%). Each tree was also put into one of five bole char categories: 0 = unburned around tree, 1 = duff burned but no bark char, 2 = light bole char, 3 = moderate bole char, and 4 = severe bole char with much bark consumption.

On November 19, 1986, all trees were surveyed for vitality, In most cases, trees were either classified as live with good vigor or clearly dead. A small percentage was initially classified as live with little chance of survival based on yellowing crowns. In the analyses, these trees were put into the dead category.

Analysis

Logistic regression models have been used in other tree damage studies to predict the probability of survival or mortality given specific tree and damage characteristics (Bevins 1980, Peterson and Arbaugh 1986, Ryan et al. 1988). Details of logistic regression theory and development can be found in Monserud (1976). The independent variables tested for value in predicting probability of survival were tree d.b.h., crown scorch class (CS), dwarf mistletoe rating (DMR), and bole char (BC). The first three are continuous variables of a quantitative nature with equal class widths while bole char is a classification or qualitative variable. The CATMOD proceedure of the SAS[3] statistical package was used for model development.

[3]The use of trade and company names is for the benefit of the reader; such use does not constitute an official endorsement or approval of any service or product by the U.S. Department of Agriculture to the exclusion of others that may be suitable.

Results

The fire damage resulted in first-year mortality of 34% of the 191 sample trees. Generally, mortality increased with decreasing tree size and with increasing crown scorch, bole char, and dwarf mistletoe infection (fig. 1). With a given fire intensity, small trees normally sustain greater damage because their crowns are closer to the heat source and their bark is thinner (Lynch 1959, Harrington 1987). In this study, 28% of trees between 3 and 12 inches d.b.h. had greater than 50% of their crowns scorched, compared to 19% of the trees greater than 24 inches d.b.h. Logically, the greater the fire damage, the greater the mortality, whether damage was crown scorch or bole char (fig. 1). There was 100% mortality for trees with greater than 87% crown scorch, whereas the most severe bole char resulted in about 67% mortality. Finally, a general trend of increasing mortality with increasing degree of mistletoe infection is apparent, with some fluctuations.

The influence dwarf mistletoe infection had on pine mortality can be inferred by comparing average pre-fire DMR of surviving and dead trees. The average DMR for trees not surviving the fire was 4.8 compared to 2.9 for surviving trees. For the entire stand, the average DMR was reduced from 3.7 to 2.9 as a result of the fire. Even though this reduction occurred in all size classes, it was more pronounced in the smallest trees. Average DMR decreased from 2.4 to 1.2 for trees between 3 and 12 inches d.b.h., from 4.2 to 3.6 for trees 12 to 24 inches d.b.h., and from 4.2 to 4.0 for trees greater than 24 inches d.b.h.

Figure 2 shows tree frequencies in various scorch class and DMR class combinations. Two trends are apparent. The percentage of trees with little or no scorch decreased as DMR increased, compared to an increase in the percent of trees with moderate to

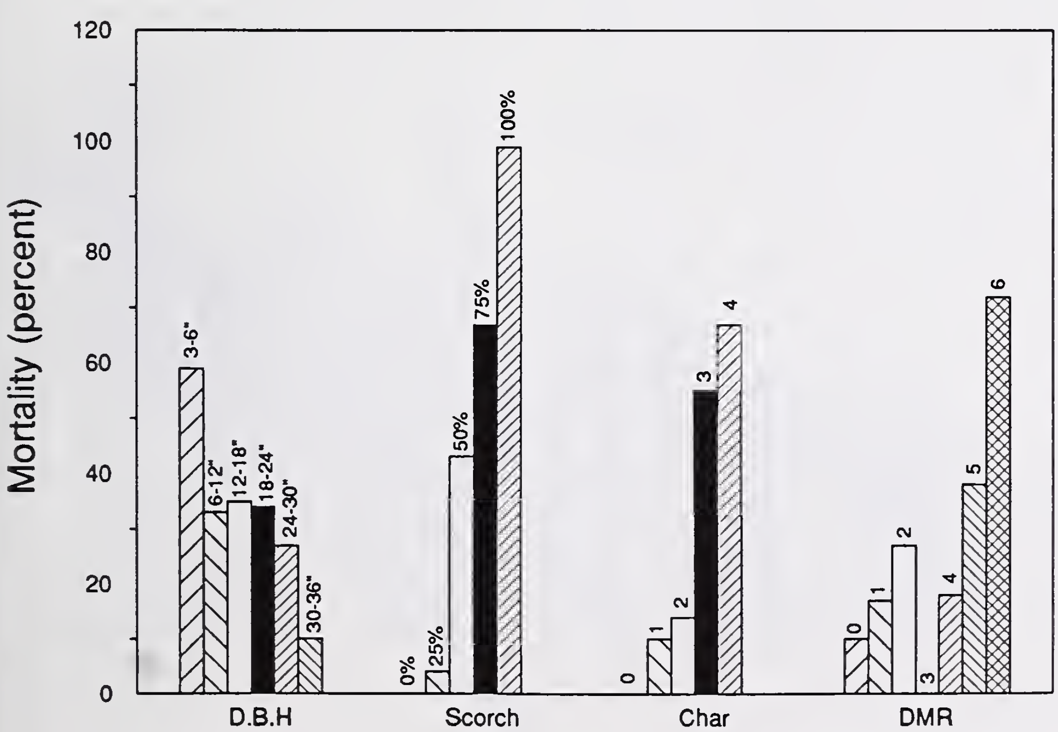

Figure 1.—Effects of d.b.h., scorch, bole char, and DMR on tree mortality. Descriptors are defined in the text.

severe scorch as DMR increased. Koonce and Roth (1985) showed that mistletoe-infected trees do not self-prune as readily as healthy trees. In their study, average height to crown bottom was 21% of infected tree height, compared to a crown bottom height of 38% for healthy trees. This implies that for a given scorch height, a greater portion of an infected tree's crown length will be scorched.

Another indicator of the relationship of scorch to DMR is shown in figure 3. The average amount of crown scorch generally increased as amount of dwarf mistletoe increased. An exception is DMR class 3, which had a sample size of only seven trees.

Mortality increased with increasing scorch, bole char, and DMR, and decreasing tree size (fig. 1). A logistic regression was developed to estimate the probability of survival using these four independent variables. The first attempt showed that bole char was not significant for predicting survival, therefore was not used. The best regression for predicting probability of tree survival was

$$Ps = 1/(1 + e^{(4.91 + 0.10\,d.b.h. - 0.10\,CS - 0.29\,DMR)}) \quad (1)$$

where Ps = probability of survival, e = base of the natural log, d.b.h. = tree diameter (inches), CS = crown scorch (percent), and DMR = dwarf mistletoe rating. The independent variables had moderate to high significance for prediction of survival; p(d.b.h.) = 0.03, p(CS) < 0.01, and p(DMR) = 0.07. Using the 50% probability level to delineate live and dead trees, the model correctly predicted vitality for about 92% of the trees. Eleven trees (5.8%) were dead when predicted to be alive, and five (2.6%) were alive when predicted to be dead.

Using the probability regression, figure 4 was generated to show the influence of scorch and dwarf mistletoe classes on probability of tree survival. The degree of crown scorch obviously has the greatest influence on survival as most trees are expected to survive within the range of the two least scorch classes, and most trees are expected to die in the greatest scorch class. DMR does influence survival within the range of the 50% and 75% scorch classes (total range is 38% to 87%). Severity of mistletoe infection is important with about 50% crown scorch of small trees, with from 50% to 75% crown scorch of medium-sized trees, and with about 75% crown scorch of large trees (fig. 4).

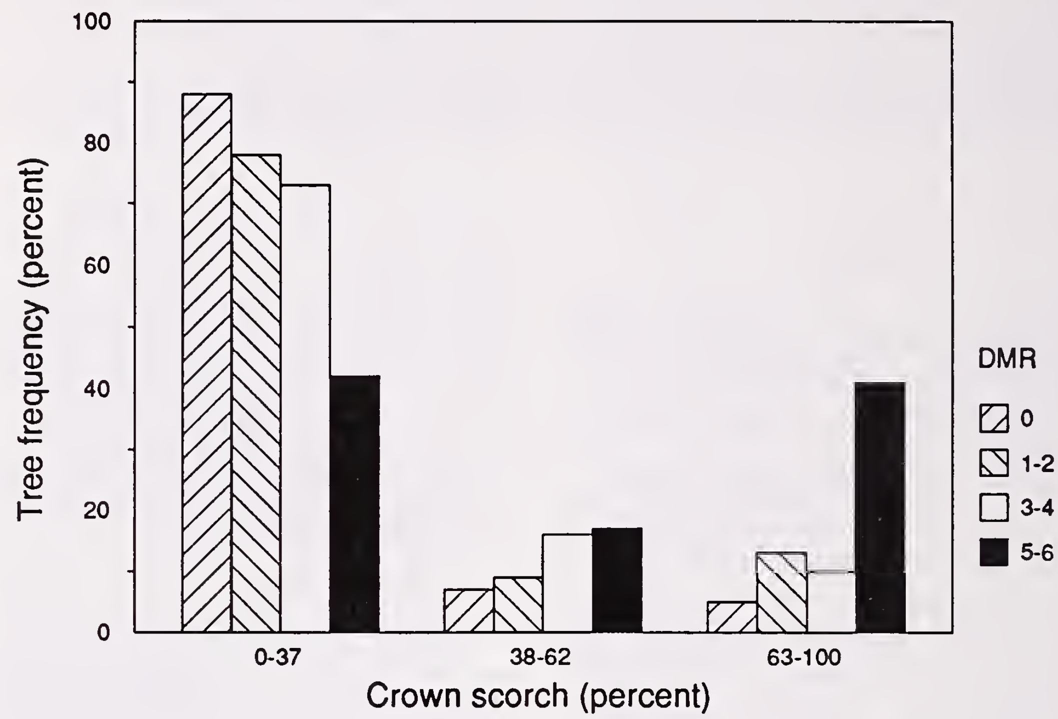

Figure 2.—Tree frequency in crown scorch classes by DMR groups.

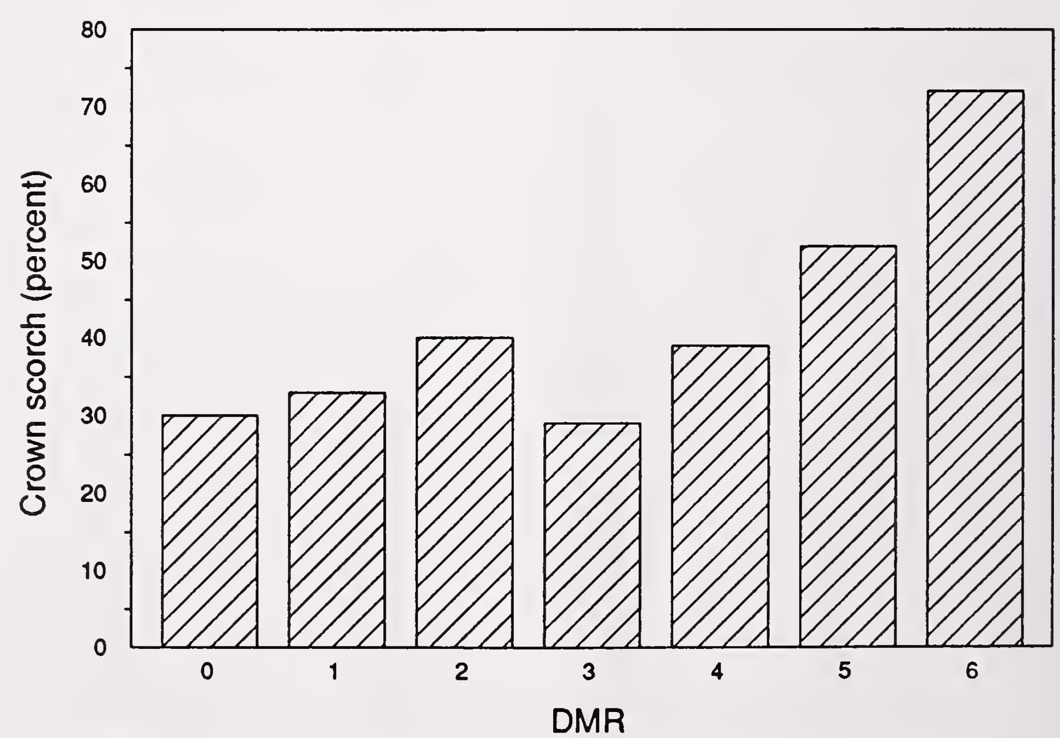

Figure 3.—Average crown scorch for each DMR class.

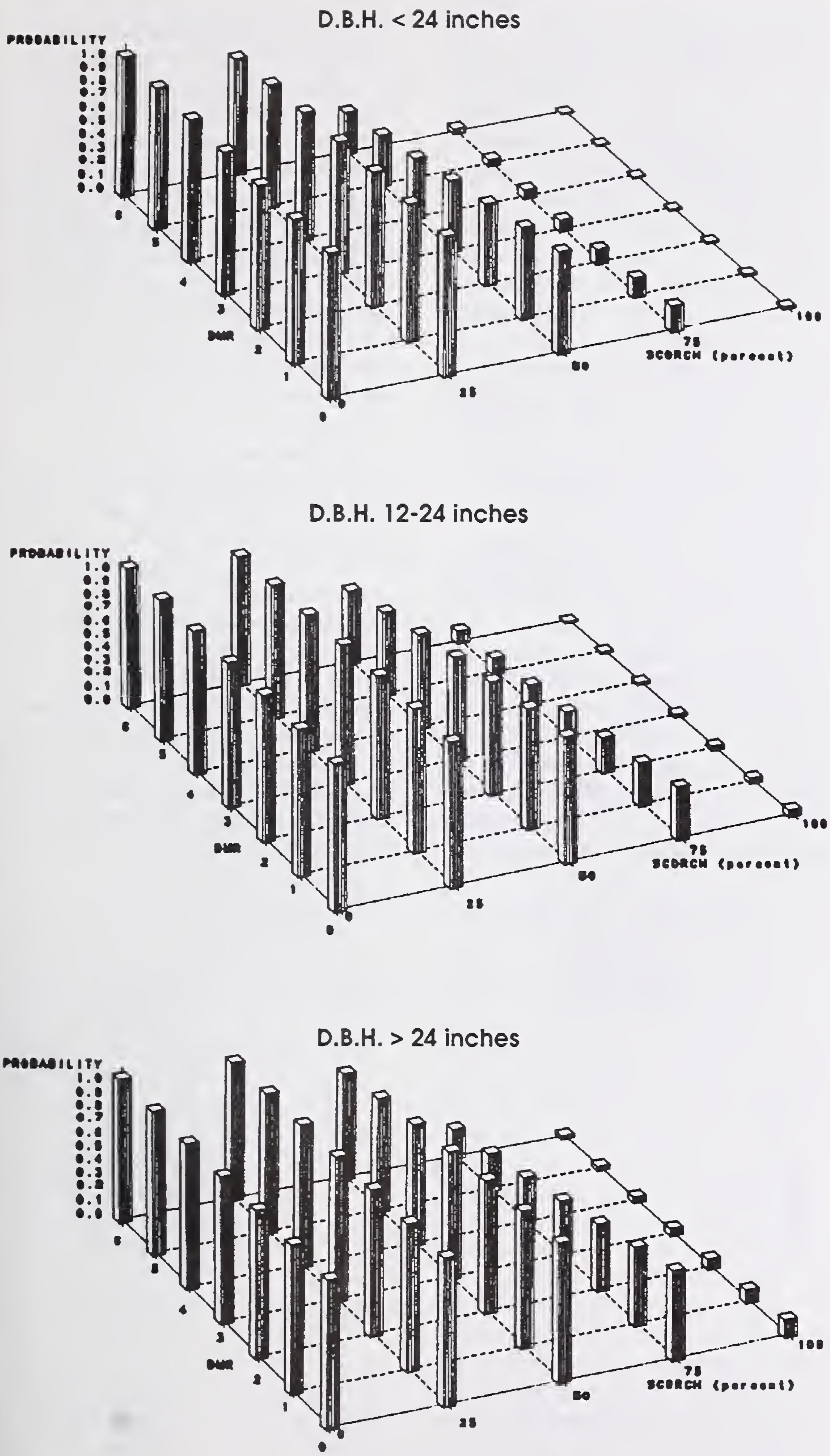

Figure 4.—Survival probability for ponderosa pine in five crown scorch classes and six DMR classes.

Figure 5 demonstrates the differences in survival probability between trees with two levels of dwarf mistletoe infection and within two key scorch classes. For an 8-inch tree with about 50% scorch, lack of mistletoe infection would lead to a 71% chance of survival, which drops to 30% for severely infected trees. Similarly, for a 30-inch tree with about 75% crown scorch, the probability of survival drops from 65% with no infection to 25% with severe infection. In other words, almost two-thirds of the large trees in good health would be expected to survive a fire damaging three-fourths of their crowns, while only about one-fourth of the trees with severe mistletoe infection could survive that same level of crown scorch.

Discussion

Dwarf mistletoe-infected trees are apparently influenced by fire in two ways. First, a larger portion of an infected tree's crown will likely be scorched with a fire of given intensity than a healthy tree's crown because of a tendency of the former to have flammable witches brooms and low crowns. Second, with equal amounts of crown scorch within the 38% to 87% range, heavily infected trees have less than half the probability of survival that uninfected trees have. This might occur simply because infected trees are less healthy, making any damage more life-threatening. In addition, if infected trees have a greater crown length than uninfected trees, similar percentage of scorching would mean a greater absolute crown volume loss in infected trees, leaving only the relatively thin, unhealthy crown tops to maintain tree vitality.

In comparison, a study in southwestern Colorado also documented crown scorch and mortality of ponderosa pine from spring, summer, and fall prescribed burns (Harrington 1987). First year mortality follow-

ing a mid-August burn in Colorado was 21% compared to 35% in our study. Several differences between study sites should be pointed out. Average diameter for trees larger than 3 inches d.b.h. was greater at the Grand Canyon site (15.6 vs. 9.5 inches). Greater general fire damage occurred at the Colorado site where 26% of the trees had 33% to 90% crown scorch compared to only 14% of the trees at the Grand Canyon. About equal percentages were scorched more than 90%. The pronounced difference probably resulting in the greater mortality at Grand Canyon, even with larger trees and less fire damage, was the widespread dwarf mistletoe infestation, which was not a factor at the Colorado site.

Key points in the discussion of dwarf mistletoe management with fire are as follows:

1. How much scorch pruning (entire branch kill) would be required for a tree or stand with a specific DMR to minimize the dwarf mistletoe infection?

2. Given the amount of scorching required, DMR, and tree size, what is the probability of tree survival?

3. If low survival is indicated, should the tree be killed and/or removed, or scorched to a lower level to improve immediate survivability?

Because dwarf mistletoe rating denotes the amount and severity of infection, it also designates that portion of the crown that should be scorch-killed for sanitation. For example, a DMR = 1 or 2 means that the lower third of the crown requires treatment; and a DMR = 5 or 6 means that the entire crown is infected, so for complete mistletoe elimination the tree would have to be killed.

An important point to consider in scorch pruning is that generally a lesser number of branches that have scorched foliage are actually killed because needles are killed easier than buds (Wagener 1961). As Ryan (1982) pointed out, species with large buds such as ponderosa pine will have bud and branch survival from a few to many feet below needle scorch depending on season of damage. Therefore, if a given percentage of scorch pruning is required, a larger percentage of the crown length needs to be scorched for the specified amount of crown to be actually killed. Therefore, to eliminate mistletoe from a tree in DMR = 1 or 2, 50% average crown scorch would be recommended. Then, knowing the amount of crown scorch needed to reduce or eliminate a specific degree of infection, an estimation of the survival probability can be made using equation [1] or figure 5.

If the probability of survival is indicated to be low, then one of two choices needs to be made based on the management objectives and silvicultural opportunities for the site. First, the tree could be killed or harvested because of poor vigor and possiblity of mistletoe dispersion. Second, the scorch level could be reduced, which would lead to an increase in probability of survival but leave some mistletoe within the residual crown.

Lightle and Hawksworth (1973) reported that there is utility in pruning severely infected lower branches from moderately to heavily infected trees. They demonstrated that trees with DMR = 4 and 5 responded to partial pruning (broom pruning) with prolonged life, vigor recovery, and growth improvement even though dwarf mistletoe was not eliminated.

A discussion of techniques for producing various degrees of scorch pruning are beyond the scope of this paper. Scorch height depends upon fireline intensity, air temperature, and windspeed (Van Wagner 1973). It can be estimated much easier using flame lengths rather than fireline intensity (Albini 1976), and can be crudely predicted using fire behavior models (Susott and Burgan 1986). In all situations where scorch-pruning is attempted, burners must be experienced in the application of prescribed fire and have good fire effects knowl-

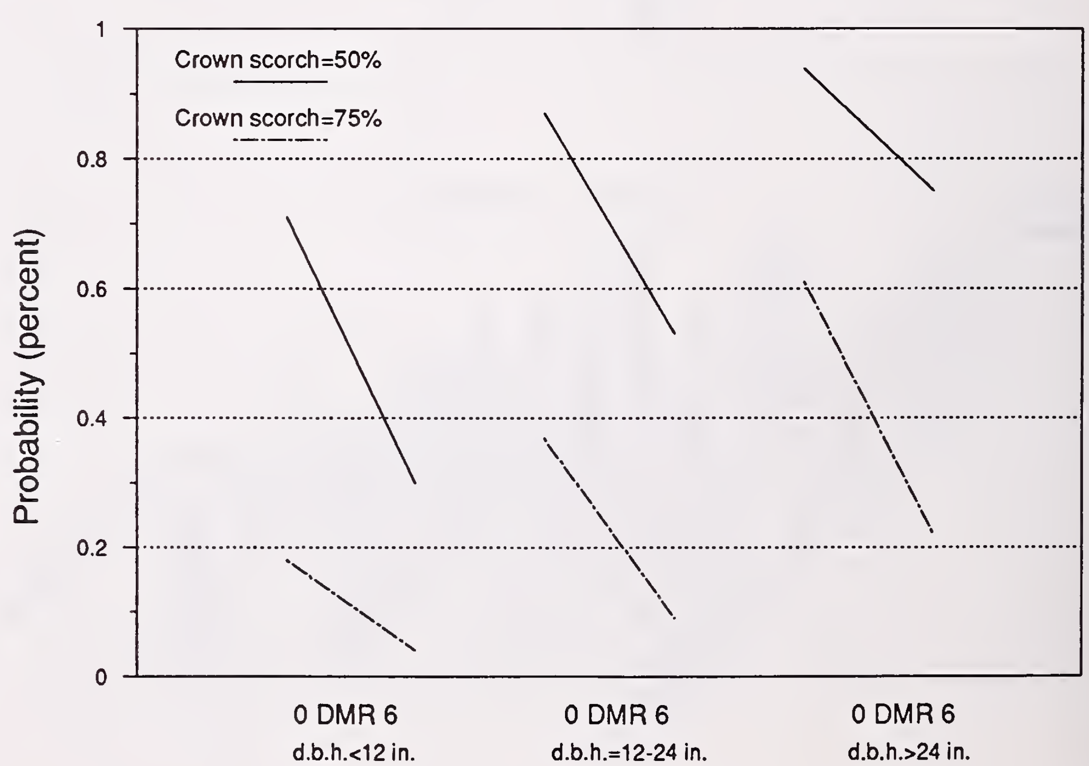

Figure 5.—Survival probability for ponderosa pine in three d.b.h. classes, two crown scorch classes, and two DMR classes.

edge to utilize proper ignition techniques (Kilgore and Curtis 1987).

The preceding discussion dealing with the potential use of fire for scorch-pruning dwarf mistletoe infected branches is based primarily on the results of this study alone. Results would likely be different if burning took place in the early spring or late fall. Therefore, the details are preliminary at best and require further investigation before wide use is recommended. However, these results show enough promise for practical application in an important area of forest management that they should be used at least as a starting point for developing guidelines for dwarf mistletoe management by prescribed fire.

Management Implications

Because presettlement fires played a major role in dwarf mistletoe abundance and distribution by eliminating severely infected dense stands (Alexander and Hawksworth 1975) and by pruning moderately infected open stands (Roth 1974), prescribed fire should not be overlooked as a possible management tool for mistletoe control. The most manageable opportunity for mistletoe control appears to be in heavily infected stands where patch clearcutting would be followed by intense broadcast burning. However, at least partial sanitation also seems possible in existing stands with light to moderate infection using specifically prescribed understory burning.

It is apparent that light crown scorching (less than 37%) will generally have little effect on tree mortality, and severe crown scorching (greater than 87%) will most often lead to death, regardless of DMR. Degree of dwarf mistletoe infection does influence survivability within the 50% to 75% crown scorch classes. This information can be used in various ways by timber, fire, range, and wildlife specialists because prescribed fires generally have multiple purposes including vegetation manipulation, hazard reduction, and site preparation.

One particular use would be in a ponderosa pine stand occupied by trees with varying severities of dwarf mistletoe. If harvesting the moderately to heavily infected trees is not feasible, a prescribed burn could be planned to produce a specified range of crown scorch such that heavily infected trees would have a low survival probability compared to uninfected trees. As an example using equation [1] or figure 5, if a group of trees averaging 8 inches d.b.h. were scorched to about the 50% level, trees with DMR $\leqslant 1$ would have about a 70% chance of survival, whereas those with DMR $\geqslant 5$ would have about a 30% chance of survival. In this study area, the disproportionate death of heavily infected trees led to an average decrease in stand DMR from 3.7 to 2.9. By exerting more control over the fire behavior under trees with various degrees of infection, the stand DMR could have been reduced further.

Another example uses this information as a guide for attempted mistletoe reduction in trees to be retained. Dwarf mistletoe in lightly infected trees (DMR = 1-3) could be reduced with about 50% crown scorch, and the chance of survival would be quite good for all size classes, 52% to 97% (fig. 5). However, if DMR = 4 to 5, then an effective amount of scorch pruning, crown scorch $\geqslant 75\%$, would result in low survival, 5% to 40%, depending on tree size and DMR. For example, an 18-inch tree with DMR = 4 would require 75% crown scorch for mistletoe control, but this would result in a 15% probability of survival. If tree survival is important because logging is not possible, cover is needed, a seed source is desired, etc., then the crown scorch could be reduced to 50%, raising the probability of survival to almost 70%, and in many cases resulting in vigor recovery.

The management of dwarf mistletoe infected stands is complex. Clearcuts and stand-eliminating prescribed fires that are often used for mistletoe control are frequently looked upon unfavorably by many influential groups. Another option, then, for mistletoe management is presented here, in preliminary form, for further testing. Applying prescribed fire to attain specific levels of crown scorch is difficult, requiring much skill and a working knowledge of analytical tools. Results will never be completely definitive because of the natural variability of forest fuels, tree responses, and fire behavior. However, with further verification of the results presented here and more knowledge gained on the effects of heat and smoke on dwarf mistletoe, management guidelines could be developed for prescribed fire use in mistletoe control.

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45

Selective Mortality With Prescribed Fire In Canyon Live Oak[1]

Timothy E. Paysen and Marcia G. Narog[2]

Abstract.—Thinned canyon live oak (*Quercus chrysolepis* Liebm.) survived prescribed understory burning despite crown and bole injury. Post burn tree mortality was 23% and was greatest in diameter classes less than 20 cm. Crown injury did not affect tree recovery. Tree recovery has continued with time. Selective thinning might be accomplished by prescribed burning.

Canyon live oak (*Quercus chrysolepis* Liebm.) is a common hardwood species in southern California. In recent years it has become necessary to actively manage this non-timber type because of its increased use by the public for fuelwood and recreation (Thornburgh, in press). Little information exists to guide management decisions for improving the quality of canyon live oak stands (Plumb and McDonald 1981).

The influence of natural fire in the evolution of many oak species (Plumb and McDonald 1981, Rouse 1986) suggests that a prescribed burning program would be feasible and desirable. Some believe that prescribed burning in oaks could be used to rejuvenate decadent or overgrown stands (Graves 1977, Hannah 1987).

Canyon live oak bark burns easily (Plumb and Gomez 1983). This may create problems during understory burning if fire carries up the trunk. However, many oak species are extremely resilient to severe stress or injury. Total foliar loss due to drought stress (McCreary 1988) or heavy bole charring due to fire (Plumb and Gomez 1983) do not always cause tree mortality.

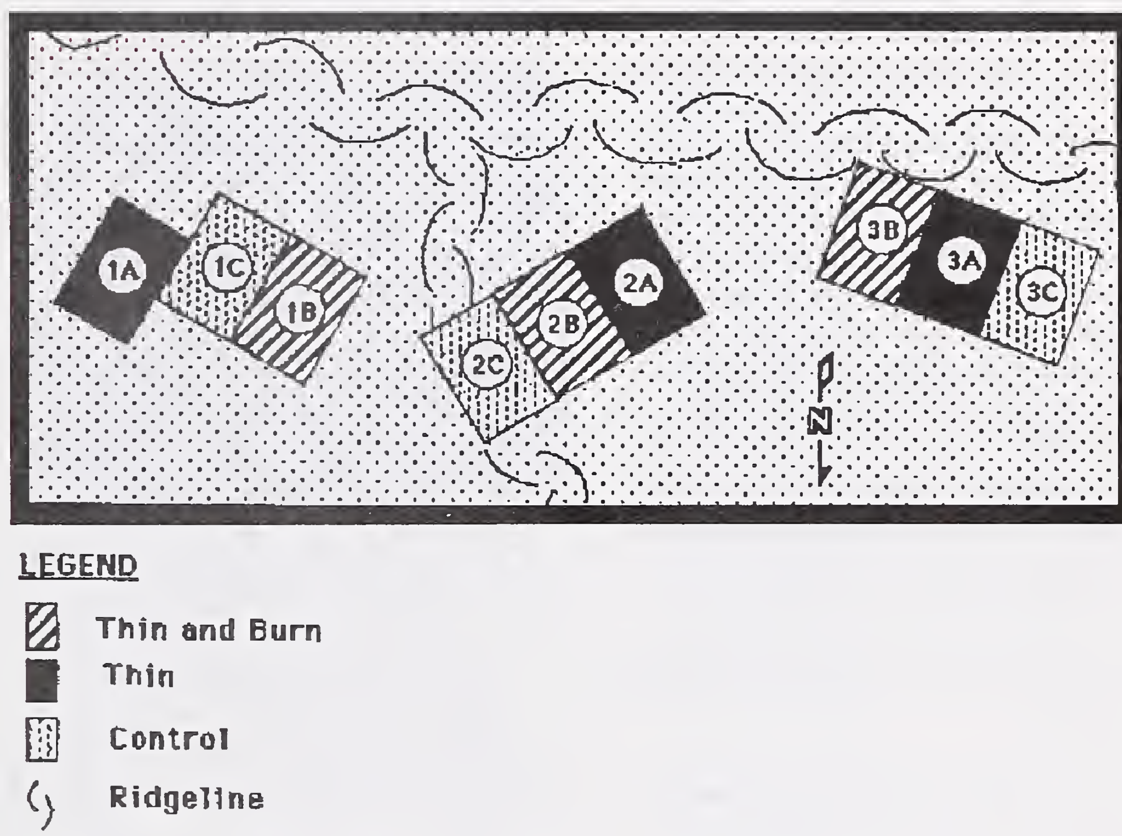

Figure 1.—Study area at Skinner Ridge, with plot layout and location of thin and burn plots 1B, 2B, and 3B.

This paper will address the first part of a larger study evaluating thinning and prescribed burning in canyon live oak to (1) improve stand quality through selective thinning, (2) reduce fire hazard through the establishment of a shaded fuel break and (3) increase diversity of wildlife habitat. We will analyze data gathered on crown recovery and tree mortality 32 months after the prescribed burn.

Methods

The study was conducted in a closed-canopy canyon live oak forest, on the north slope of Skinner Ridge (elev. 1500 m), on the San Bernardino National Forest in southern California. Aspect varied slightly among plots (fig. 1), and slope ranged from quite steep (35°) and convex to moderate and concave. Litter varied in depth, but was relatively complete over all plots. Wildfire burned the site about 100 years ago.

[1] *Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2] *Research Forester and Ecologist, Forest Fire Laboratory, Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture, Riverside, CA.*

The study area was divided into three blocks, each containing three plots which measured 30 m by 40 m. Within each block, three treatments, control, thin, and thin and burn, were randomly assigned to the plots. The oaks were thinned from a stand basal area of 54 m^2/ha to 22.5 m^2/ha. Thinning was as equitable as possible across diameter size classes, and between single stemmed trees and stems from multiple-stemmed clusters. Stand thinning, completed in June 1985, took 15 months. The prescribed burn treatment was accomplished during November 1985.

Three months after the burn, mortality, diameter, bole injury, and percent live crown were recorded for all oaks with stems inside the perimeter of each thin and burn plot. Four 3-m-wide strips, which were horizontal to slope, were positioned every tenth meter in each plot (fig. 2), and were used as subsamples for subsequent data collections in May 1987 and August 1988.

We assessed above ground stem mortality of the oaks based on crown and bole condition. Root survival was not considered a criterion for tree survival, even though canyon live oak commonly resprout from the root area.

Percent crown vitality was visually estimated for each tree, based on live foliage observed on branches and twigs. Complete crown mortality was assigned if no green foliage was apparent in the tree crown. The crown was designated as 100% live if no scorched foliage or foliage loss was evident. Estimates were for all percentages of live crown between these two extremes.

Bole damage was evaluated by a visual inspection of the bark and cambium. A bark scraper was used to assess damaged tissue. Obvious color or texture deviations from those observed in living oak tissue were considered to be evidence of fire injury. Classification of injury was based on the percent circumference damaged by the fire.

Results

Flames did not reach the canopy so consumption did not occur in the tree crowns. But, heat convection from the fire did cause extensive foliar injury in most trees. Initial assessment of injury showed that bole and crown damage did not always occur together or to the same degree. Over 98% of 635 oaks on the thin and burn plots sustained some injury from the fire. Ninety-eight percent of the trees had some degree of crown injury, while obvious bole damage was found in only 64% of the trees.

Table 1.—Mean percent (with standard deviation in parentheses) live canopy found in thin and burn plots.

Plot	1986	1987	1988
I	27(38)	40(31)	44(35)
II	17(27)	28(24)	37(34)
III	28(36)	40(26)	60(27)
All	24(34)	37(28)	47(37)

In January 1986, mean crown damage per tree (limited to foliar death) from the fire was 76%. Foliar death of the crowns continued, and by spring 1986 all trees or portions of trees within the perimeter of the burn appeared to be dead. In May 1987, a subsample of the burned plots showed 37% mean crown recovery

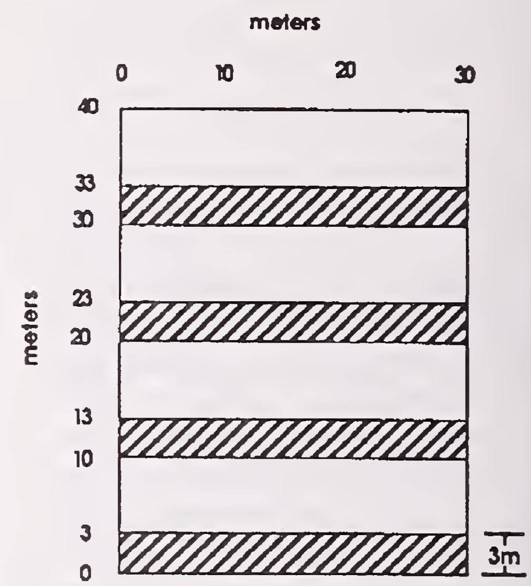

Figure 2.—Subsample layout for thin and burn plots (1B, 2B, 3B).

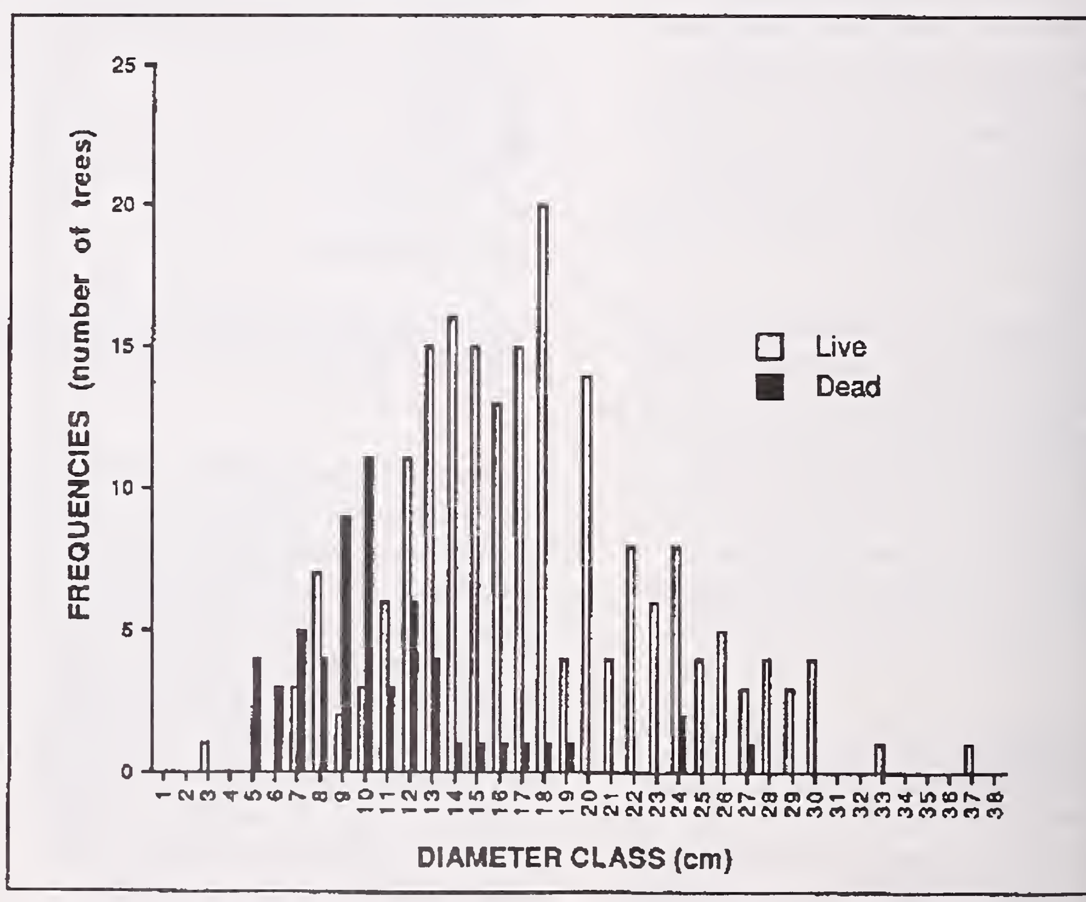

Figure 3.—Live and dead DIAMETER frequencies for all plots.

per tree as evidenced by refoliation. By August 1988, overall recovery for the three plots had reached 47% (table 1).

After 32 months, overall tree mortality was 23%. Sixty-nine percent of all bole injured trees showed some heat injury on 50% to 100% of the bole circumference, within a vertical range of 4 m from the ground. With only a few exceptions on plot 1B, the oaks that died had smaller diameters than those that survived (fig. 3). Plot one was steeper and deeper layers of slash had accumulated around tree bases. The diameter of the live vs. dead trees constituted two distinct populations. According to the Rank-Sum Test this difference was highly significant (right-handed tail probability = 0.9999).

Conclusions

The preliminary results presented in this paper suggest that prescribed understory burning probably can be used to manage canyon live oak. After 32 months, post burn mortality of trees was low—23%, even though fire injury was sustained by most trees—98%. Greater mortality occurred in trees smaller than 20 cm diameter, indicating that understory burning may prove valuable for selective thinning. Mortality of the few larger diameter trees probably resulted from long residence times of burning around the base of the trees. In general, canyon live oak was able to withstand understory burning despite bole and foliar injury.

Continued monitoring in this canyon live oak stand is necessary before a final decision can be made regarding improvement of stand quality after burning. Plumb and Gomez (1983) found that oak crown mortality can occur as many as 8 years after a fire. Post fire effects, yet to be analyzed, include possible changes in tree growth rate or pattern, susceptibility to pests and diseases, and vigor of remaining trees.

Acknowledgments

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Appropriate Suppression Response on the Gila National Forest[1]

Stephan H. Servis and Janet F. Hurley[2]

Abstract.—The Forest Service revised its fire policy in 1979. Each wildfire ignition requires a timely suppression response with appropriate forces based upon established fire management direction and cost efficiency. Comparison of similar fire seasons shows that the current policy of appropriate suppression response has reduced per acre costs of fire suppression by over 50%.

Prior to the revision of the fire policy in 1979, the Forest Service direction was to control wildfires at ten acres or less by 10 a.m. the next day. This is called the 10 a.m. policy.

The 10 a.m. policy was revised to the present one, requiring a timely response for each wildfire ignition. Each fire is managed using a suppression strategy based upon local conditions of terrain and weather, established fire management direction, and cost efficiency. Suppression strategies direction may range from direct control, with minimum acreage burned, to more indirect methods of containment and confinement. In all cases, suppression strategy is designed to avoid unacceptable resource losses.

The Gila National Forest is using the appropriate suppression response not only within the wilderness, but throughout the entire Forest. All three suppression strategies are being used and are based on the Forest's Land and Resource Management Plan direction.

Discussion

The Forest first implemented the revised policy in 1985. In doing so we have gained much support from various publics. However, there has been no evaluation to date of the cost effectiveness of the revised policy. We will answer this by comparing similar fire seasons under both policies, with a case study of a 1986 fire.

The 1985 fire season (appropriate suppression response policy) was compared to the 1974 fire season (10 a.m. policy). These two seasons were very similar in fire weather and total numbers of fires. The 1974 Fire Fighting Fund (FFF) dollars were inflated to the 1985 values so we could compare average per acre costs for the two policies.

	1974 10 a.m. policy	1985 Appropriate suppression response
Total fires	556	406
Total acres	31,013	22,805
Total FFF	$3,920,075	$1,416,478
Aver. cost per acre	$126	$62

Thus, the comparison of the two seasons shows a reduction of over 50% in the average per acre cost of fire suppression.

Case Study: the Granite Fire

The Granite Peak area of the Gila Wilderness has had 10 fires since 1909, with some parts burning four times in the last 79 years. All of these fires have been caused by lightning and occurred during the fire season.

During the 1986 fire season, the Granite fire was discovered on April 17, at 1605 MT. The District Fire Management Officer flew the fire to observe fire behavior and topographic characteristics. The fire was creeping down slope from the ridgetop, and natural barriers, including previous burns, existed to confine the fire to 1500 acres.

These two factors influenced the decision made under the Escaped Fire Situation Analysis (EFSA) to use the appropriate response of confinement, even though winds were 30 to 35 mph.

On May 8, the strategy was changed to containment. The change in response was due to continued high winds, increasing daytime temperatures, low humidity, and a forecast for these conditions to continue. The fire was also approaching the natural barriers. Twelve firefighters were then used to ensure that this fire remained contained within the natural barriers. The suppression cost was $3000, and the final size was 1350 acres.

Had this same fire occurred under the old 10 a.m. policy, the following would have taken place. We would have dispatched a minimum of two crews and used retardant. These crews would have had to be flown in with a helicopter and supported either with the helicopter or pack stock.

Estimated time to control and mop up the Granite Fire at 40 acres would be three days. Estimated costs, based on Incident Command System 209 forms are as follows:

[1] *Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2] *Servis is Forest Fire Management Officer, U.S. Forest Service, Gila National Forest, Silver City, NM.; Hurley, District Ranger, Wilderness Ranger District, Gila National Forest, Silver City, NM.*

Two crews @ 3200/day for 2 days	= $6400
One crew @ 3200/day for 1 day	= $3200
Two airtankers four trips	= $3146
Four loads of retardant @ 1400/load	= $5600
One lead plane	= $416
Five helitack crewpersons @ $255/day for 3 days (until fire was out)	= $3825
Flight time on Helicopter 308 @ $228/hr. for 10 hrs.	= $2280
Total	= $24,867

In this scenario, we would have controlled the fire at 40 acres for a cost of $24,867. The use of appropriate suppression response for this fire realized a saving of $21,867. This is just one of many examples where the appropriate suppression response has saved money.

Conclusion

Review of the last 4 years of appropriate suppression response on the Gila National Forest shows that the policy is cost-effective, while preventing unacceptable resource losses. Implementing this policy when fire behavior can be extreme tested the skills and knowledge of our fire managers. Lightning-caused fires are increasingly allowed to return to their natural role in the wilderness, under the prescribed natural fire program. These past burns, together with the EFSA process, have given our managers the tools to utilize the appropriate suppression response strategies.

Feedback Mechanism in a Chaparral Watershed Following Wildfire[1]

Burchard H. Heede[2]

Abstract.—As chaparral regrows after wildfire, buffer strips develop and accumulate enormous volumes of sediment. Future destruction of these strips by fire or other agents would make the deposits easily available for transport. Thus, the processes of restoration initiate a negative feedback mechanism if research and management do not find ways to curb this situation.

The Problem

Natural recovery of vegetation following wildfire in chaparral watersheds leads to changes in microtopography. Because chaparral does not regrow uniformly, a mosaic pattern results. This regrowth often forms barriers to sediment delivery from uphill bare sites (Heede 1988). Soils derived from coarse-grained granites, which contain very little binding materials are highly erodible. Thus, relatively large volumes of sediment are deposited at the uphill edge of the vegetation barriers, or buffer strips. Since these mounds and other ground surface undulations remain uncompacted for relatively long time spans, at least three decades in our case, their stability depends on the soundness of the buffer strips.

If an intense storm follows a wildfire, another type of sedimentation occurs in the channel network. Because deep, loose and coarse sediment deposits favor subsurface flows, most surface flows are small and immense volumes of sediments are deposited in the channels.

The objectives of this paper are to identify sediment processes on the watershed and in the channels following wildfire in chaparral, to project future developments, and to discuss management implications.

Past Work

Much has been written about wildfires in chaparral watersheds, as well as the catastrophic consequences. But to this writer's knowledge, none of the studies spanned relatively long time periods—two decades or more. Thus, immediate fire effects were the focus of these investigations. In contrast, Heede et al. (1988) reported on sediment delivery linkages in a chaparral watershed covering a period of 26 years. They showed that postfire developments are complex and can only be interpreted if both the watershed and the channel network are investigated. The essence of their findings was that the watershed and the channel network were not in the same geomorphological stage. While sediment delivery from the watershed to the stream channels had ceased, sediment movements continued within the channel network.

Study Area

El Oso Creek watershed (drainage area, 2.5 km^2) is located in central Arizona on the east flank of the Mazatzal Mountains. Average elevation is 1100 m; bedrock geology consists predominantly of deeply weathered coarse-textured Precambrian granite. In spite of relatively high annual precipitation (677 mm), the climate must be considered semiarid. Only 33% of the annual precipitation falls in summer when extremely high temperatures (reaching 43° C) reduce the effective precipitation substantially. The vegetation cover consists of chaparral that has regrown since an intensive wildfire 29 years ago. Today, about 95% of the original plant canopy has been restored.

El Oso Creek is an ephemeral stream. Following the wildfire, immense amounts of sediment were deposited in the channels during subsequent intense storms. Although the total deposits were estimated at 2.5×10^6 m^3, based on seismographic investigations,[3] it is believed that part of this amount originated from earlier fires. In the main channel, fills up to 25 m were found.

Methods

Prefire and sequential postfire aerial photographs were used to determine qualitatively the development

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Research Hydrologist, Rocky Mountain Forest and Range Experiment Station, Forestry Sciences Laboratory, Arizona State University Campus, Tempe, AZ 85287-1304.*

[3]*Laird, J. R.; Harvey, M. D. 1986. Complex-response of a drainage basin to geomorphologically-effective fire. Tempe, AZ: U.S.Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 199 p. (unpublished report).*

of vegetation and erosion conditions. Relative chaparral densities could be obtained. The loci and extent of erosion and deposition as well as their changes with time were also determined. Latest developments were verified on the ground.

Ten microwatersheds, ranging in size from 0.01 to 0.2 ha, were selected to represent a range of vegetation conditions on different lithologies, elevations, and slope angles for the measurement of overland flow and sediment delivery (fig. 1). These microwatersheds also have different ground cover characteristics. The term microwatershed was used, because they are larger in size than those used in traditional plot studies, and the boundaries consisted of natural overland flow divides wherever possible. Where divides were not sufficiently pronounced, artificial watershed boundaries were created using sheet metal strips sunken into the ground surface.

Ground cover characteristics were represented by erosion pavement (3 microwatersheds) and erosion pavement with a vegetation buffer strip uphill from the measuring station (3 microwatersheds) (fig. 2). The buffer strips consisted of relatively dense chaparral stands, 2.5 to 4.0 m wide.

Overland flow and sediment were caught by 4-m-long sheet metal troughs (fig. 2). Sheet metal strips at each side of the troughs assured this catch. Collector tanks made possible volumetric determinations of flows and sediment concentrations.

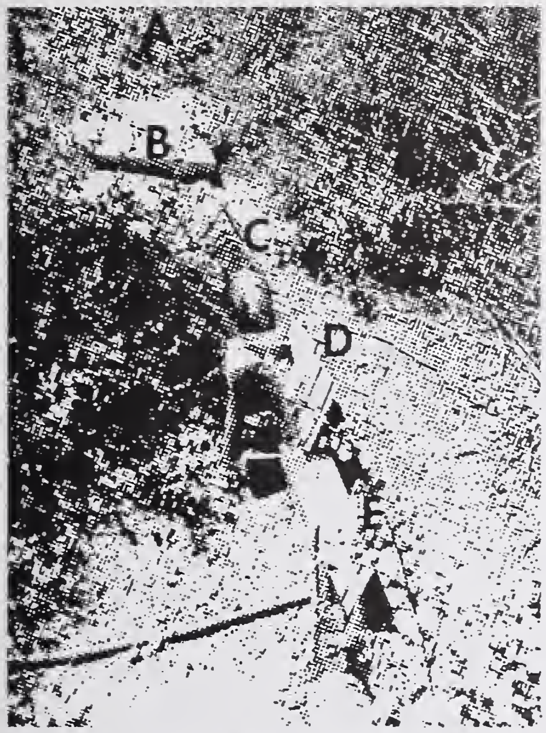

Figure 1.—Uphill view of part of an installation below a chaparral buffer strip; (A) chaparral buffer strip; (B) collector trough for overland flow; (C) conveyance pipe; (D) supercritical flume; (E) flume for intake to sediment pumping sampler. Pipe in foreground leads to a collector tank.

Results

Judging from the first postfire photographs, storms following the intense wildfire must have led to increased overland flows and immense volumes of sediment delivered into the stream network. This postfire behavior is demonstrated by our field research on erosion pavements (table 1). Now, nearly three decades after the fire, overland flow from bare sites (erosion pavements) still averages more than 100 times that from chaparral buffer strips. It must be assumed that immediately after the fire, this difference was considerably larger. DeBano (1966) has shown wildfires induce hydrophobic soil conditions, which cause non-wettability of the soils and increase overland flow and sediment transport.

On our bare microwatersheds, sediment delivery was on the average over 300 times larger than on bare microwatersheds with buffer strips (table 1). Due to the hydrophobic soil properties immediately following the fire, sediment deliveries during the early postfire storms also must have been much larger than those of today's bare areas.

Increased infiltration into soils under the buffer strips greatly reduced overland flow and caused substantial accumulations of sediment uphill from the strips. Deposit depths up to 0.45 m were measured (fig. 3). With future depth increases, water withdrawal by the loose, coarse, granite-derived sediments also will increase, and increasingly stronger overland flows will be required to move the sediment.

As the first postfire photographs show, the tributary channels were clogged by sediment and the main channel lost depth and widened con-


Figure 2.—Looking across a microwatershed of erosion pavement with chaparral buffer strip at the downhill border.

siderably. Apparently, stream competence was not sufficient to move the incoming material through the system; thus deposition occurred. Today, practically all channel banks are lined by chaparral buffer strips. Even south aspects along El Oso Creek developed strips of substantial width (15 to 25 m). Greater soil depth and higher soil moisture seem to favor this development.

Now that sediment movement from the slopes has largely stabilized, relatively clear water reaches the channels. When clear waters enter the clogged tributary channels, the available free water energies begin to move the sediment. Channel scour has started in the headwaters and is proceeding downstream. Flow entering the main channel is absorbed by the deep and porous sediment deposits, and the incoming sediment is deposited as in-channel fans (fig. 4). These fans are still growing due to lack of surficial flows with sufficient transport capacity. As more deposition occurs, the storage sites widen, surficial flow depth decreases, and even more sediment is stored.

Conclusion

The localized sediment accumulations behind buffer strips represent an enormous reservoir of easily available sediments, should a wildfire strike the watershed in the future. This possible chain of events constitutes a negative feedback mechanism as follows:

buffer strip loss ↘
massive erosion ↙
regrowth of strips ↖
sediment depositions ↗

Table 1.—Average annual sediment delivery and average annual overland flow from the two different types of microwatersheds based on 3 years of data (data in parentheses represent standard errors).

Microwatershed Type	No.	Sediment delivery *kg ha⁻¹ year⁻¹*	Overland flow *mm year⁻¹*
Erosion pavement	3	6,404	3.7
	4	1,521	39.1
	5	329	20.8
	Av.	2,751	21.2
		(1,859)	(10.2)
Erosion pavement with buffer strip	6	5.28	0.1
	7	5.94	.5
	8	17.09	0
	Av.	9.44	.2
		(3.83)	(.1)

Figure 3.—Excavation of the loose sediment deposits uphill from a buffer strip. The deposits increase in depth downhill. Approximate depth of this excavation is 0.40 m. Retractable tape measure (arrow) in the excavation denotes scale.

Figure 4.—An in-channel fan of sediment formed by flows from a tributary. Long arrow denotes direction of tributary flow. Short arrow shows flow direction in the main channel of El Oso Creek.

Thus, discouragingly, the restoration of a burned chaparral watershed sets the stage for the next catastrophic event, should an unusual storm follow a fire.

During three postfire decades, sediment redepositions in the stream network have not led to channel restoration, except in the headwater reaches of the tributaries. Large volumes of sediment are still ready for transport by an exceptional flow. Hence, long-term instability characterizes the main channel.

Management Implications

This research suggests that conducting relatively frequent prescribed fires with low intensities could reduce the rates and time frames at which sediment is delivered to the channels. This would reduce the likelihood that an intense chaparral wildfire would radically alter stream system morphology by the movement of large volumes of sediment. Carefully conducted prescribed fires could in many cases exclude burning of the buffer strips lining the channels, thus further reducing impacts on the stream. Certainly, additional research is needed to study the consequences of increasing the frequency of prescribed fire on the chaparral associated erosional processes.

Although channel structures could be considered where downstream values dictate control of sediment transport from the watershed, they would be very expensive.

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Prescribed Fire in Arizona Ponderosa Pine Forests: A 24-Year Case Study[1]

Peter F. Ffolliott and D. Phillip Guertin[2]

Abstract.—A prescribed fire was set to consume three-fourths of the forest floor depth in a ponderosa pine forest. Evaluations of the effects of this prescribed fire were made 1 month, and 1, 2, 11, and 24 years after the fire. The objective of the fire was accomplished. Other effects of the fire included thinning of the forest overstory from below, increased ponderosa pine seedling establishment, increased production of herbaceous plants, and a temporary reduction of fire hazard. The future of prescribed burning in Arizona's ponderosa pine forests seems favorable.

A prescribed fire was set to burn approximately three-fourths of the forest floor depth in a ponderosa pine forest near Flagstaff, Arizona, in October 1964. The forest floor, by definition, is the accumulation of dead organic plant material on mineral soil. This objective of the prescribed fire generally was achieved (Davis et al. 1968). Other effects of the fire included the thinning of the forest overstory from below, an increase in the germination and initial survival of ponderosa pine seedlings, and a small increase in the production of herbaceous plants. A temporary reduction of fire hazard also was attained.

The study described in this paper was conducted to analyze the effects of this fire through time after the burning, on those characteristics originally modified by the burn. In essence, this paper presents a case study of the effects of prescribed fire in a ponderosa pine forest, as evaluations of these effects were made 1 month, and 1, 2, 11, and 24 years after the fire. A partial insight into the future of prescribed burning in these forests also has been obtained from this study.

Methods

Two one-fourth-acre areas, designated Area A and Area B, with similar characteristics were selected for burning (table 1). On each area, 16 sample points were spaced systematically, 25 by 25 feet. Measurements taken at each of the sample points included:

Forest floor depth.—Total depth measured before and 2, 11, and 24 years after the prescribed fire.

Needle deposition.—Needle drop caught by a 12- by 12-inch hardware cloth square in burned and unburned conditions before and 1 month, and 1 and 2 years after burning. These measurements were discontinued 2 years after burning.

Fire effects on trees.—Crown damage of trees tallied by point sampling with a 25-factor angle gage was classified as (1) severe—more than two-thirds damaged, (2) moderate—one-third to two-thirds damaged, (3) light—less than one-third damaged, and (4) none—no apparent crown damage. Mortality in each crown damage class was recorded 2 years after the fire.

Forest density.—Square feet of basal area per acre estimated by point sampling with a 25-factor angle gage. Diameter breast high (dbh), crown position, and length of live crown were recorded for all trees tallied. Measurements were taken before and 2, 11, and 24 years after burning.

Table 1.—Characteristics of areas selected for prescribed fire (Davis et al. 1968)

Characteristics	Area A	Area B
Forest floor		
Depth (inches)	1.7	3.0
Weight (tons per acre)	10.2	17.6
Forest overstory	Ponderosa pine poles, 5 to 11 inches dbh, with scattered sawtimber and a few Gambel oak	
Basal area (ft² per acre)	170	305
Herbaceous plants	Dominately perennial grasses, including muttongrass and bottlebrush squirreltail	
Soils	Volcanic	Volcanic
Topography	Level terrain	30% to 35% slope to the southeast

[1]*Poster paper presented at the conference, Effects of Fire in Management of Southwestern Natural Resources (Tucson, AZ, November 14-17, 1988).*

[2]*Professor and Assistant Professor, School of Renewable Natural Resources, College of Agriculture, University of Arizona, Tucson, Arizona 85721.*

Seedling germination and survival.—Mil-acres stocked, 16 on and 16 adjacent to each burned area. Observations were made 1, 11, and 24 years after burning.

Herbage production.—Weight of grasses, forbs, and browse species estimated on 9.6-square-foot plots before and 1, 11, and 24 years after burning.

To achieve the objective of consuming three-fourths of the forest floor depth, a moderately intense surface fire, with flame heights generally 1 to 2 feet, was prescribed. The conditions selected for the prescribed fire and those observed on the two study areas are presented in table 2.

Although the areas burned were relatively small, it was concluded that the same burning procedure apparently would have been successful on larger areas (Davis et al. 1968). The fuels ignited easily and carried the fire well. Estimated fireline intensities were 48 BTUs per second per foot on Area A, and 90 BTUs per second per foot on Area B. In retrospect, these estimated fireline intensities probably were high. Flame heights of about 2 feet generally are associated with fireline intensities of 30 to 35 BTUs per second per foot (Martin et al. 1979). The fire on Area B was more intense than on Area A because of the slope, and higher wind speed and air temperature at the time of burning. The ignition lines on Area B were spaced more closely than on Area A to hold the flame heights to the prescribed limits.

Results and Discussion

Forest Floor

The prescribed fire consumed 71% of the forest floor depth on Area A, and 73% of the depth on Area B (Davis et al. 1968). On the combined areas burned, the L layer, consisting of unaltered organic material, was consumed completely at 69% of the sample plots. The F layer, consisting of partly decomposed organic material, and the H layer, consisting of well-decomposed organic material, were consumed completely at 13% of the sample plots. It was concluded, therefore, that the objective of setting a prescribed fire to consume three-fourths of the forest floor depth was satisfied.

Eleven years after the prescribed fire, the depth of the forest floor on Area A was 0.8 inch, or 47% of the pre-fire depth of 1.7 inches (Ffolliott et al. 1976, 1977). On Area B, the forest floor depth was 1.2 inches, which was 40% of the original depth. In the 11 years since the fire, the additional needle fall that had accumulated on the two burned areas was 15 to 20% of the pre-fire forest floor depth.

Twenty-four years after burning, the forest floor depth on Area A was 1.1 inches, or nearly two-thirds of the pre-fire depth. The depth of the forest floor on Area B was 2.2 inches, or 73% of the original depth. Needle fall and subsequent forest floor development on the two areas in the 24 years since the prescribed fire represented 35% to 45% of the pre-fire forest floor depth.

The forest floor on the two burned areas has been returning to pre-fire conditions, at least in terms of depth. Furthermore, in the 24 years since the prescribed fire, the depth of the forest floor on the two areas has been approaching that generally found in cutover ponderosa pine forests in north-central Arizona (Ffolliott et al. 1968) and in the Southwest (Sackett 1979). The density of the forest floor on the burned areas remains less than that in unburned ponderosa pine forests, however.

An unknown quantity of fire-killed twigs, branches, and small trees have fallen to the ground on both burned areas in the 24 years since the fire.

Needle Deposition

Immediately following the prescribed fire, about the same amount of needles fell on both the burned areas and adjacent unburned areas (Davis et al. 1968). The period of this accumulation represented the normal needle-drop period for Arizona's ponderosa pine forests. During the remainder of the first year after burning, however, the needle deposition on the two burned areas was greater than on the unburned areas (table 3).

Needle deposition decreased on the burned areas 2 years after burning, apparently in response to the decreased volume of tree crowns. Needle fall on the adjacent unburned areas, when "prorated" to account

Table 2.—Conditions selected for prescribed fire and observed conditions when fires were set (Davis et al. 1968).

Burning conditions	Prescribed fire	Area A	Area B
Fuel moisture			
L and F layers	6-12%	8.6%	8.0%
H layer	15% or more	17.6	26.0
Fuel temperature			
Upper 1 inch	80° F., average	86° F., sun 75° F., shade	85° F., sun 75° F., shade
Air Temperature	75° F. or higher	75° F.	80° F.
Wind velocity in flame zone	2-5 mph	1-4 mph	3-5 mph
Weather	Clear	Clear	Clear
Ignition pattern	Strips into the wind or downslope, spaced 10-20 feet to maintain prescribed flame height		

for differences in time, showed little change in comparison to the previous depositions.

As previously mentioned, the measurements of needle deposition were discontinued 2 years after the prescribed fire.

Tree Mortality

Trees killed by the fire, or damaged and subsequently died, generally were the suppressed and intermediate saplings (less than 5 inches dbh) (Davis et al. 1968). Mortality was less in poles (5 to 11 inches dbh) and sawtimber (greater than 12 inches dbh). No trees less than 4.5 feet in height survived on either of the burned areas. Seventy-six percent of the severely damaged trees died within 2 years of the fire, a similar result to that reported in a fire near the Fort Valley Experimental Forest (Herman 1950). Most of the moderately and lightly damaged trees survived.

Forest Density

A general effect of the prescribed fire was a thinning from below (Davis et al. 1968). On Area A, 47% of the pre-fire basal area was lost in 2 years, a reduction to 90 square feet per acre. Twenty-three percent of the original basal area was lost on Area B, a reduction to 235 square feet per acre. Again, on both areas, most of the basal area reduction was in suppressed and intermediate saplings. Area A lost more of its pre-fire basal area because a greater proportion of the trees were saplings, which were more susceptible to damage by the fire than the generally larger trees (in the pole and sawtimber size classes) on Area B.

Eleven years after burning, the basal area on Area A had increased to 120 square feet per acre, indicating an average growth rate of 3% annually (Ffolliott et al. 1976, Ffolliott et al. 1977). However, on Area B, the basal area was 210 square feet per acre, a forest density level that was less than that measured 2 years after the prescribed fire. Apparently, enough trees initially damaged by the fire had died in the intervening time period to offset the growth of the residual trees.

There has been little change in the basal areas on the burned areas in recent years. Twenty-four years after the fire, the basal area on Area A was 127 square feet per acre, an increase of only 6% in the last 13 years. On Area B, the basal area was 213 square feet per acre, which represented an increase of less than 1 percent. It appeared that, on both areas, growth has been offset largely by mortality in recent years. Both of the burned areas still contain too many trees for maximum wood production (Schubert 1974), so the growth rates remain low, relatively.

Seedling Germination and Survival

More ponderosa pine seedlings germinated and survived initially on the burned areas than on adjacent unburned areas (Davis et al. 1968). One year after the fire, newly started seedlings occupied 85% of the mil-acre plots on Area A and 95% on Area B, compared with 20% and 12%, respectively, of the same number of unburned plots. Most of these new seedlings were short-lived, however.

Seedlings established since the fire stocked only 25% of the mil-acre plots on both of the burned areas 11 years after the fire (Ffolliott et al. 1976, Ffolliott et al. 1977). None of the unburned plots supported seedlings at this time.

Twenty-four years after the prescribed fire, established seedlings still occupied 25% of the mil-acre plots on Area A. However, no plots on Area B supported seedlings, and no seedlings stocked the unburned plots adjacent to either burned area. The failure of seedlings to survive on the burned areas has been attributed largely to the returning of the forest floor to pre-fire conditions.

Herbage Production

Annual herbage production on Area A increased from 3 pounds per acre before the fire to 40 pounds per acre 1 year after the burn (Davis et al. 1968). Most of this increase was attributed to the presence of mullein, a relatively unpalatable forb. Herbage production on Area B had remained at the pre-fire level of 5 pounds per acre. It was concluded that grazing values, which were negligible before the prescribed fire, were not changed by burning.

Eleven years after the fire, annual

Table 3.—Needle deposition on the burned areas and adjacent unburned areas (Davis et al. 1968).

	Area A		Area B	
Post-fire period	Burned	Unburned	Burned	Unburned
	tons per acre			
First 34 days	0.72	0.73	0.80	0.76
Remainder of first year	1.75	.96	2.72	.78
Second year	.76	1.76	1.58	2.19
Total	3.23	3.45	5.10	3.73
Difference	-.22		+1.37	

herbage production on Area A also was approximately 40 pounds per acre, although the species composition had changed (Ffolliott et al. 1976, Ffolliott et al. 1977). Mullein had been replaced by a mixture of bottlebrush squirreltail, mutton bluegrass, showy goldeneye, red-and-yellow-pea, and buckbrush. On Area B, the herbage production had increased to 17 pounds per acre, and the species composition generally was similar to that on Area A. Still, the grazing values were considered negligible on the burned areas.

Annual herbage production had decreased on both burned areas 24 years after burning. On Area A, the herbage production was 13 pounds per acre, while that on Area B was 11 pounds per acre. These decreases in herbage production appeared to be related, in part, to the general return of the forest floor to pre-fire conditions. Earlier work in Arizona's ponderosa pine forests had shown that herbage production decreased as the total depth of a forest floor increased (Clary et al. 1968), as reported here.

Fire Hazard

The fire hazard on the burned areas was reduced by the consumption of nearly three-fourths of the forest floor. However, this reduction was only temporary, as the L layer of the forest floor was built up to pre-fire levels in 2 years (Davis et al. 1968). In addition, an unknown quantity of fire-killed limbs and small trees fell to the ground in the years immediately after the fire (Ffolliott et al. 1976, Ffolliott et al. 1977). If the areas in question had been burned again at regular intervals, the fire hazard might have remained low.

Management Implications

The future of prescribed fire in Arizona's ponderosa pine forests seems favorable. As survivors of periodic wildfire (Dieterich 1980), these forests should benefit from carefully prescribed burning programs. Prescribed fire already is used to dispose of debris from logging operations and to thin dense forest stands, with roughly 35,000 to 40,000 acres of debris burned annually by the USDA Forest Service. The use of fire to reduce unwanted herbaceous vegetation and small trees requires more skill in selecting the conditions for burning, and probably not more than 6,000 to 8,000 acres of National Forest land are treated in this way in Arizona each year.

Through the removal of dead organic material on mineral soil, other potential benefits of prescribed fire can include increased seedling establishment and reduced fire hazard. Herbage production also can be increased, although the burning should be prescribed on more productive sites or in lower density forest stands than reported upon herein. Importantly, many of these benefits only will be temporary, unless prescribed fire is scheduled at regular intervals.

Obstacles in the way of prescribed fire programs are a lack of people experienced in prescribed burning, a difficulty in protecting forests from "runaway" fires in areas where fuels (including logging debris and herbaceous plants) have been allowed to accumulate, and insufficient funding for the program. Fortunately, these are not insurmountable obstacles, and progress is being made toward the use of prescribed fire as an effective tool in forest management.

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Rocky Mountain Forest and Range Experiment Station

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